# Operational interactions between marine mammals and commercial fisheries in Australian and South Pacific waters: characterisation and options for mitigation.

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December 2012

Thesis submitted for the degree of Doctor of Philosophy

THE UNIVERSITY of ADELAIDE

SCHOOL OF EARTH AND ENVIRONMENTAL SCIENCES

*For Kate, my love.*

*Freedom is the absence of choice.*

Pol Pot

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# Thesis abstract

Reports of interactions between marine mammals and fisheries are on the increase globally. This is mainly because fishery effort has increased to feed the burgeoning human population and because advances in technology have allowed fisheries to exploit habitats that were until recently inaccessible. Additionally, many marine mammal populations decimated by harvesting over the past few hundred years are recovering and the growing conservation community is paying unprecedented attention to their welfare and conservation generally, with growing interest in their interactions with fisheries.

Operational interactions are conspicuous and involve the close contact of marine mammals with fishing gear, either because marine mammals opportunistically or habitually target fishing activities to depredate (i.e. attempt to consume) caught fish, or because marine mammals incidentally encounter fishing gear while foraging naturally. Operational interactions often result in negative outcomes for the conservation and welfare of the marine mammals involved and for the economic viability of the fisheries involved. Marine mammals that become by-caught may receive life threatening injuries from entanglements, or may drown, thus having adverse impacts on small or recovering populations. Fisheries that are targeted by depredating marine mammals may need to replace damaged fishing gear, or may have the catch partially or completely removed, thus having adverse impacts on their economic viability.

At the time this body of work commenced, little was being done to address the known or suspected occurrence of operational interactions between marine mammals and several commercial fisheries in the Oceania region. The general aim was to make significant inroads into addressing thesis, by:

1. Reviewing a major fishing method in the two regions in which there are operational interactions with marine mammals;
2. Characterising the nature and extent of depredation and by-catch where operational interactions are known to exist; and
3. Where deemed necessary in those fisheries, developing mitigation strategies and explore their efficacy.

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Collectively, the five research chapters in this thesis address these aims. They are stand alone case studies of marine mammal depredation and by-catch in commercial fisheries, four of which have already been published in international, peer reviewed journals. The first three research chapters focus on operational interactions involving odontocetes (i.e. toothed whales) and the second two research chapters focus on the otariids (i.e. eared seals).

Chapter 2 generally defines and reviews the nature and extend of odontocete (i.e. toothed whale) depredation and by-catch in longline fisheries, which has emerged as an environmental and economic concern internationally. At least 20 odontocete species are involved across all major oceans, although depredation and by-catch rates were variable. This study also introduces fishing gear modification as a viable mitigation strategy. Chapter 3 builds on this theme in more detail by exploring depredation and by-catch, mainly by pilot whales (*Globicephala* spp.), false killer whales (*Pseudorca crassidens*) and melon headed whales (*Peponocephala electra*), in pelagic longline fisheries targeting tuna in Australia and Fiji. Two devices were developed to physically or psychologically deter depredating odontocetes. Unfortunately, the rarity of depredation and by-catch events did not enable the efficacy of the devices to be properly assessed, although both were found to be easily integrated into the normal fishing practice and to have little or no impact on target fish catch rates. Chapter 4 attempts to specifically address the efforts of a purse-seine fishery operating in South Australia (SA) in reducing by-catch of common dolphins (*Delphinus delphis*), pursuant to conditions set out under the Australian Government *Environment Protection Biodiversity Conservation Act 1999* (EPBC Act). After characterising the nature and extent of the problem, it was found that a combination of a Code of Practice (CoP) using avoidance and release strategies and of gear modifications resulted in a reduction in encirclement by-catch from an estimated 377 to eight mortalities each year.

Chapters 5 and 6 attempt to assess the impact of a demersal gill-net fishery on the *Endangered* Australian sea lion (*Neophoca cinerea*) in waters adjacent to SA. Specifically, Chapter 5 assesses the performance of the Great Australian Bight Marine Park (GABMP) in protecting animals of populations residing within it, pursuant to a management plan that aims to uphold the spirit of the EPBC Act. It was found that individuals tracked using satellite transmitters spent only 27.7% of their time inside the GABMP and could travel up to 9 times further than the location of the southern boundary. Additionally, it was found that by-catch occurred beyond the southern boundary and also within the

GABMP during the six months each year that the fishery was allowed to operate within it, with an estimated 14 to 33 individuals killed each 17.6 month breeding cycle. Based on these findings, Improvements to the GABMP were recommended. In a similar manner to chapter 4, chapter 6 directly addresses recommendations pursuant to the EPBC Act to quantify the impact of a demersal gill-netting on all Australian sea lions across SA, by quantifying the extent of geographic overlap and the level of by-catch mortality and extent of overlap between the two. It was found that the two overlapped extensively in 68.7% of 4 $km^{-2}$ cells and that by-catch was high, at 283 to 333 killed each breeding cycle. Based on these results, it was suggested that a network of permanent and temporary closures along with more extensive monitoring of fishing activities be considered.

In summary, this thesis demonstrates that with sufficient political will, stakeholder support and the necessary funds, by-catch and depredation issues can be addressed and can lead to favourable outcomes for the marine mammal populations and commercial fisheries involved. Each case study presented provides many lessons, some being specific to the operational interaction, the marine mammal species or the fishery, and some being more generally applicable. Regarding the latter, more general lessons, it is acknowledged that depredation and by-catch are statistically rare events that may vary across time and space. As such, investigating and addressing them is likely to be costly, with the results still only providing a snapshot or a broad estimate that may not be representative of the overall problem. Additionally, marine mammals are intelligent and may quickly learn how to circumvent mitigation measures, despite their complexity and cost. Although marine protected areas (MPAs) such as the GABMP are implemented with the best intentions, they are often of insufficient size to provide adequate protection and may also allow a level of fishing that still has a quantifiable impact.

Despite these drawbacks, all stakeholders are encouraged to adopt a spirit of collaboration and of commitment to attempting to resolve operational interactions with marine mammals where possible. Acoustic deterrence devices have many problems that are yet to be resolved, including their impractically large size and limited sound propagation and battery life. Nonetheless, their amalgamation with some of the physical deterrence technologies developed in chapter 3 may provide a more generic method of deterrence across all fisheries, thus providing hope that resolving operational interactions between marine mammals and commercial fisheries may be a viable proposition in the future.

## Personal acknowledgements

Following tradition, firstly I would like to sincerely thank my supervisors for assisting me along the path towards achieving this life changing goal. Peter Shaughnessy was always available to provide endless and balanced advice, and proved to be an extremely tolerant sounding board for all of my ideas, the good ones and the not so good ones. Peter reviewed and improved endless drafts, all the time allowing me to take my own direction. He cared deeply about all aspects of the work and remained concerned for my welfare, especially during the more trying periods. Peter had the skill of putting me back on track with simple, insightful statements; the quintessential 'optim-o-realist'. Tim Ward stepped in at a time when all looked lost in the early days and helped me to embrace a broader approach to the project than I had originally intended. Despite the extra work, this turned out to be a very good idea. Tim also taught me some key lessons in writing that helped me to prepare the first publication associated with the thesis. I will benefit from those lessons throughout my career. Simon Goldsworthy facilitated my first academic opportunity; to explore the operational interaction between Australian sea lions and demersal gill-netting. I learned through my operational interactions with Simon that I should stick to my principals, whatever the outcome. Without doubt, this approach helped me to (perhaps stupidly) expand the project and to carry on to the bitter end, no matter how trying the conditions; kill or be killed.

Among my friends and colleagues, I must first thank Simon Childerhouse, my first marine mammal science 'boss'. Aside from being a 'doer' and collegial king, Simon also enthusiastically adopted the unofficial role of PhD supervisor, helping me to remain focused on finishing my thesis after I had returned to work. Aside from providing endless assistance with developing ideas, organizing logistics and funding, then structuring and editing drafts, Simon provided incredible motivation in a sports coach like manner. Simon singlehandedly dragged me over the finish line; you would not be reading any of these words had it not been for him. Included among the fonder memories from my PhD days will be gaining Simon as a lifelong friend.

Now to my fellow PhD stooges. Al Baylis's contribution has to be among the most diffuse and difficult to identify, although doubtless influential. So far as I can remember, Al provided many philosophical gems during the earlier years that helped crystallize my thoughts at numerous impasses. Not surprisingly though, most only became apparent the next day, among the fussiness of a dull headache and gin breath, over a much needed cooked breakfast. I will forever miss the stress relieving motorcycle and Land Rover journeys mate; trans Asia awaits. Completing the small sphere of good blokes are Luke Einoder, Chris Izzo and Kris Peters. Collectively and as individuals, your friendship, advice, humour, antics and company kept me sane and helped me enjoy my time in Adelaide, a place that offered me little else. From here, I very much look forward to our continued friendships.

*

Several others also deserve my sincere thanks, making the bad smell of my PhD considerably more reminiscent of fresh cow dung on a spring morning, rather than dog excrement stuck to the bottom of ones shoe on a hot day. Throughout the whole thing, Guy Abel and Phil Exton provided me with unconditional friendship and support. During the earlier period, Guy and his mother Lesley graciously provided much needed shelter at a time when a cardboard box or stormwater drain were realistic alternatives. Both Phil and Guy suffered my various complaints like troopers and still managed to provide so much encouragement. Unforgettable are the numerous times you both dragged me out of the office to enjoy the sun, sand and surf on days when PhD guilt would otherwise have resulted in me languishing unproductively in dimly lit and air conditioned discomfort. Simon Clark enabled me to survive the darker days in Port Lincoln by being an ally and a true friend, regularly reminding me to take life a little less seriously. I will always remember with a smile his natural powers of persuasion during sea lion captures. Simon Jarman provided me with news so good that I literally collapsed with happiness, at a time when Malaria had me by the throat and the purveyors of evil were about to repossess my house. Ever since, Simon and Andrea Polanowski, the latter being my fabulous office mate, have both kept me real with many philosophical gems and truisms, often over a delicious single malt or micro brew. Both of you provided the subtle encouragement I needed to slowly steer my PhD ship out of the storm. Hayden Dyke, the one I have known the longest, provided me with surveyor like ground truthing as only a true friend could. All the while his influence, friendship and advice have continually reminded me that life isn’t all about PhDs. Among Hayden’s many pearls of wisdom was perhaps the most important lessen, which he has unwaveringly set by example over the years: respect is earned, experience requires a committed journey and neither are an instantaneous entitlement.

Finally, to my loved ones. Liam, this collection of words is my gift to you. Watching you grow (mostly from afar) from a child into a young man demonstrates that this period of distraction took much too long to pass. Mum, Dad, John and Iris, your constant faith in me has been humbling and provides the greatest proof that kinship is the strongest force in life. I truly regret that Dad, the one who nurtured my love of the sea from the beginning, didn’t make it this far to see the fruits of my (and his) labour. And here I conclude by mentioning my dearest love Kate. Consider the chronology of these acknowledgements akin to leaving the crescendo till last in a classic thriller novel, or the most exquisitely prepared piece of gravy covered pork crackling till last while eating a fine Sunday roast. All those clichés about love apply to you in infinite abundance; you *are* my axis, the centre of my universe. You have been the greatest positive influence and guiding light of all, providing me with *everything* so selflessly at a time when I have been completely selfish: love, advice, support, understanding, awareness, encouragement, faith, food… the long list goes on.  Most of all you have given me a past, present and future full of love and hope.  I am nothing without you, so I dedicate this thesis to you babe.

## Statement of originality

This work contains no material which has been accepted for the award of any other degree or diploma in any university or other tertiary institution to Derek J Hamer and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in the text.

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Name: Derek John Hamer

Signature:

Date: 27 May 2013

## Associated publications

During my PhD candidature at the University of Adelaide, my aim was to prepare all of the case studies as chapters, to a standard worthy of journal publication. When the time came to compile the thesis, four of the five chapters had been published in international, peer reviewed journals (one being featured), while the remaining chapter was published as an Australian Government report. They are, in order of their appearance in this thesis:

**Hamer, D.J.**, Childerhouse, S.J., Gales, N.J., 2012. Odontocete by-catch and depredation in longline fisheries: a review of the literature and of potential solutions. *Marine Mammal Science* 28, E345-E374 (Chapter 2)

**Hamer, D.J.**, Childerhouse S.J., 2012. Physical and psychological deterrence strategies for mitigating odontocete by-catch and depredation in pelagic longline fisheries: progress report. Progress report to Pacific Island Forum Fisheries Agency (FFA), Worldwide Fund for Nature (WWF) South Pacific, and Pacific Islands Tuna Industry Association (PITIA). Australian Marine Mammal Centre (AMMC), Department of Sustainability, Environment, Water, Population and Communities (DSEWPaC). 47pp. (Chapter 3)

**Hamer, D.J.**, Ward, T.M., McGarvey, R., 2008. Measurement, management and mitigation of operational interactions between the South Australian Sardine Fishery and short-beaked common dolphins (*Delphinus delphis*). *Biological Conservation* 141, 2865-2878. (Chapter 4)

**Hamer, D.J.**, Ward, T.M., Shaughnessy, P.D., Clark, S.R., 2011. Assessing the effectiveness of the Great Australian Bight Marine Park in protecting the endangered Australian sea lion (*Neophoca cinerea*) from by-catch mortality in shark gill-nets. *Endangered Species Research* 14, 203-216. (Chapter 5)

**Hamer, D.J.**, Goldsworthy, S.D., Costa, D.P., Fowler, S.L., Page, B., Sumner, M.D., 2012. The endangered Australian sea lion regularly becomes by-catch in and extensively overlaps with demersal shark gill-nets in South Australia. *Biological Conservation*. DOI: 10.1016/j.biocon.2012.07.010 (Chapter 6)

Over the same period, I was also involved in the publication of several other papers. Those most relevant of those to this thesis are as follows:

**Hamer, D.J.**, Grayson, J.  (eds). 2012. Issues paper for the Australian sea lion (*Neophoca cinerea*). September 2012. Report for the Threatened Species Scientific Committee (TSSC), Department of Sustainability, Environment, Water, Population and Communities (DSEWPaC). 50pp.

Izzo, C., **Hamer, D.J.**, Bertozzi, T., Donnellan, S.C., Gillanders, B.M., 2011. Telomere length and age in pinnipeds: the endangered Australian sea lion as a case study. *Marine Mammal Science* 27, 841-851.

Lowther, A.D., Harcourt, R.G., **Hamer, D.J.**, Goldsworthy, S.D., 2011. Creatures of habit: foraging habit fidelity of adult female Australian sea lions. *Marine Ecology Progress Series* 443, 249-263.

Shaughnessy, P.D., Goldsworthy, S.D., **Hamer, D.J.**, Page, B., McIntosh, R.R., 2011. Australian sea lions *Neophoca cinerea* at colonies in South Australia: distribution, abundance and trends, 2004 to 2008. *Endangered Species Research* 13, 87-98.

Baylis, A.M.M., **Hamer, D.J.**, Nichols, P., 2009. Assessing the use of milk fatty acids to infer the diet of the Australian sea lion (*Neophoca cinerea*). *Wildlife Research* 36, 169-176.

**Hamer, D.J.**, Ward, T.M., McGarvey, R., 2009. Objective reporting of scientific results is critical for maintaining relationships with industry and achieving conservation outcomes for fisheries. *Animal Conservation* 12, 287-288.

## Statement of co-authorship

All five research chapters in my PhD thesis have been published (four in international, peer reviewed scientific journals and one as an Australian Government report). The completion of each was made possible through the contribution of many others, who have been either formally acknowledged, or have been included as co-authors. Co-authorship was offered to those who ultimately assisted by providing opportunities, data, advice and expertise. Ultimately though, most of the fieldwork and analysis, and all of the writing was completed by me. Naturally, the degree and nature of the contribution made by each co-author varied considerably, although sincere gratitude is extended to all involved. Each co-author has signed below, indicating their endorsement of the work and of co-authorship. They are, in alphabetical order:

Childerhouse, Simon J.

Clark, Simon R.

Costa, Dan P.

Fowler, Shannon L.

Gales, Nick J.

Goldsworthy, Simon D.

McGarvey, Rick

Page, Brad

Shaughnessy, Peter D.

Sumner, Mike D.

Ward, Tim M.

# General introduction: operational interactions between marine mammals and commercial fisheries

This thesis is presented as a series of five main stand alone research chapters, or case studies, that explore the nature and extent of operational interactions between marine mammals and commercial fisheries, with a view to either developing and testing, or suggesting, mitigation strategies. Four of the five chapters have been published or accepted for publication in international, peer reviewed journals, while the other has been published as an Australian Government report. Following, the introductory chapter sets the scene by describing the background, context and detail of the problem and of potential solutions. At the end of the thesis, the discussion chapter attempts to summarise and make sense of the key findings presented in each research chapter and to provide a meaningful synthesis.

## 1.1 Background

"*Conflicts, real or imagined, between fishing and marine mammals are long-standing*"; this simplistic statement appeared in one of the earliest seminal works dedicated to exploring the relationship between marine mammals and commercial fisheries (Beddington et al. 1985). This message remains relevant today, nearly three decades on. Mitigating these conflicts, or 'interactions' as they have become more widely known, has proven difficult. This is because they are diverse and complex in nature, thus making it difficult to identify, develop and test appropriate and practical solutions. However, as the frequency and rate of reported interactions between marine mammals and fisheries continues to increase (due either to a greater level overlap between the two, or to an increased desire or capacity to report), the imperative to implement mitigating strategies has become increasingly important.

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The first serious attempt to address tis issue occurred in 1981, when the International Union for the Conservation of Nature (IUCN), with assistance from the United Nations (UN) Food and Agriculture Organization (FAO) and the UN Environment Program (UNEP) project, convened a *Workshop on Marine Mammal/Fishery Interactions* in La Jolla, California. The attending experts concluded that interactions between marine mammals and commercial fisheries were (i) geographically widespread, (ii) involved most types of fishing gear and many populations and species of marine mammal, (iii) was probably detrimental to both and (iv) remained largely misunderstood and unaddressed (Beverton 1985; Hofman and Bonner 1985). The emerging literature indicates that otariids (i.e. one of the two groups of pinnipeds, often referred to as the eared seals) and odontocetes (i.e. the toothed whales) are the groups predominantly involved (Woodley and Lavigne 1991; Culik, 2004). Nonetheless, successful attempts to address and mitigate the problem have struggled to pace with the considerable increase in reported incidence of operational interactions between marine mammals and commercial fisheries over the past two to three decades (e.g. Northridge 1984; Wickens 1995; Culik 2004; Read 2008; Hamer et al. 2012).

## 1.2 Context & definition

### *1.2.1 Trophic & operational interactions: the distinction*

In general terms, the nature of marine mammal and fishery interactions can be divided into two broad categories: 'trophic competition' and 'operational interactions'. Trophic competition, sometimes referred to as ecological or cryptic interactions, occur when marine mammal individuals, populations or species compete with a fishery for access to the same finite fish stock (Beverton 1985; Northridge and Hofman 1999; Plaganyi and Butterworth 2005). For the purposes of this introduction chapter, the term 'fish' is used in a very general sense, primarily although not exclusively referring to the bony fishes (Osteichthyes), elasmobranches (sharks *Selachii*, rays and skates *Batoidea*) and the squids (Teuthida), because they are the three groups most often targeted by commercial fisheries. Trophic competition is unlikely to be observed and is difficult to quantify directly, because its occurrence is geographically widespread (potentially in all areas where the fishery, the marine mammal and the mutually targeted fish stock coexist) and temporally protracted (cyclic or constant, over days, seasons or years; Plaganyi and Butterworth 2005).

Most trophic relationships are likely to be highly complex, involving many elements (e.g. encounter rates, diet, prey availability, population dynamics and multi-species interactions) and fluctuations (e.g. regularly by season, or irregularly). This makes it difficult to determine the severity and consequences of the impact that commercial fishing activity may have on marine mammal populations, or that marine mammals may have on commercial fishing (Trites et al. 1997; Bowen et al. 2002; Matthiopoulos et al. 2008). Additionally, competition may be occurring at different levels in the food web, where a fishery may harvest fish species that are the prey of fish species consumed by marine mammal (Trites et al. 1997). For example, Cape fur seal (*Arctocephalus pusillus pusillus*) predation of shallow-water cape hake (*Merluccius capensis*) was predicted to result in greater recruitment of commercially targeted deep-water cape hake (*Merluccius paradoxus*), because adults of the former readily consume the larvae of the latter (Punt and Butterworth 1995). Furthermore, the consequences of trophic interactions may take considerable time to become apparent. For example, killer whale (*Orcinus orca*) predation in the northeast Pacific Ocean was hypothesised to have shifted from baleen whales when they were harvested to commercial extinction during the 1960s and 1970s, to pinnipeds, then to sea otters (*Enhydra lutris*; Estes et al. 2004). The subsequent collapse of the sea otter population during the early 1990s resulted in the explosion of sea urchin (Echinoidea) populations and the rapid increase in benthic barren grounds where kelp beds had once flourished. These examples indicate that trophic competition is difficult to identify and quantify in time and space and may also have broader, indirect impacts that appear unrelated. Additionally, managing the effects of competition with a fishery on marine mammal populations has generally been confined to the use of blunt instruments, such as reducing quota limits, or implementing temporal or spatial closures, the performance of which are typically difficult to assess. Therefore, characterising and quantifying trophic competition can be difficult, and managing it can be problematic and costly.

In contrast, operational interactions, which are sometimes referred to as direct or overt interactions, have a conspicuous and observable component. As such, they are comparatively easy to quantify and can thus have the potential to be effectively managed, because the performance of mitigation strategies can be observed and assessed. In their simplest form, operational interactions occur when marine mammals and fishing gear come into close or 'physical' contact (Northridge and Hofman 1999; Shaughnessy et al. 2003; Read 2005). Although the net or line caught fish may be part of the natural diet of the marine mammal involved, operational interactions may also occur where trophic competitions do not occur. For example, a marine mammal may come into physical contact with fishing gear while in pursuit of free swimming natural prey fish, or may opportunistically feed on fish caught in fishing gear that would otherwise be inaccessible under natural foraging circumstances (e.g. Forney and Kobayashi 2007; Hamer and Childerhouse 2012). Whatever the case, this type of interaction typically results in an immediate and negative consequence, either by the marine mammal becoming entangled, injured or drowning, or by the fishing operation loosing fish or hauling damaged fish (Bonner 1982; Beverton, 1985; Wickens et al. 1992; Northridge and Hofman 1999; Read 2005). A detailed description of these consequences is provided in the following section.

Operational interactions may be one of the most pressing anthropogenic threats to marine mammals today (Read 2008; Hamer et al. 2012). The relative ease with which operational interactions can be identified, characterised and quantified, when compared with the interactions associated with trophic competition, suggests that identifying, testing and implementing mitigation strategies for operational interactions should also be a tractable proposition. Therefore, the general focus of this thesis is to identify and characterise cases of operational interactions between marine mammals and commercial fisheries in the Oceania region and then, where appropriate and possible, explore, develop and test potential mitigation strategies.

### *1.2.2 By-catch & depredation: their nature & impact*

Two readily observable elements of operational interactions have attracted the most interest from a management perspective. They are 'by-catch', when a marine mammal becomes caught in the fishing gear, and 'depredation', when fish caught in fishing gear are damaged or removed by a marine mammal (Northridge and Hofman 1999; Reeves et al. 2001; Read 2005). Given the intense and ubiquitous nature of today's commercial fishing activities in coastal and offshore marine environments (FAO 2009), it is not surprising that the majority of otariid and odontocete species have been or are currently reported to be involved in one or the other, or both of these activities (Northridge 1991; Woodley and Lavigne 1991; Culik 2004; Hamer et al. 2012).

Marine mammal by-catch occurs in all major fishing methods. Examples including large odontocetes (e.g. killer whales) and mysticetes (e.g. North Atlantic right whales *Eubalaena glacialis*) in gill-nets (Read 2008), smaller odontocetes (e.g. spotted dolphins *Stenella attenuata*) in purse-seine nets (Gosliner 1999), small (e.g. common dolphin *Delphinus delphis*) and large odontocetes (e.g. false killer whale *Pseudorca crassidens*) in longlines (Gilman et al. 2006; Hamer et al. 2012) and phocids (e.g. harp seals *Phoca groenlandica*) and otariids (e.g. New Zealand sea lions *Phocarctos hookeri* and Australian fur seals *Arctocephalus pusillus doriferus*) in trawl nets (Pemberton et al. 1994; Wilkinson et al. 2003; Hamer and Goldsworthy 2006). By-catch is also geographically ubiquitous, occurring in all major oceans (Woodley and Lavigne 1991; Culik 2004;

Hamer et al. 2012). Despite acknowledgement of this widespread problem among fishers, fishery and conservation managers, there is a paucity of information on the impact of by-catch on most of the marine mammal populations involved. Among those reports available, four serve as good examples. Firstly, there is concern about the conservation of the false killer whale population that has operational (and possibly trophic) interactions with and becomes by-catch in the Hawaiian pelagic longline fishery (Reeves et al. 2009), and secondly, the freshwater baiji (*Lipotes vexillifer*) may have gone extinct due, in part, to by-catch in a variety of fisheries in the Yangtze River, China (Turvey et al. 2007). Thirdly, by-catch or harbor porpoises (*Phocoena phocoena*) in the Baltic region may be causing the already depleted population to decline further, suggesting stated conservation objectives are not being met (Berggren et al. 2002), and fourthly, by-catch of Hector's dolphins (*Cephalorhynchus hectori*) is likely to have caused population decline along the east coast of New Zealand's South Island (Slooten and Davies 2012).

The impact of by-catch may be greater than vessel-based records are able to reveal. It is known that some by-caught individuals escape alive with material entangled around their neck (typically the case with otariids; e.g. Page et al. 2004) or around their tail or fluke (typically the case with cetaceans; e.g. Vanderlaan et al. 2011). These events are likely to lead to associated injuries, infections and starvation, which may eventually cause death (Best et al. 2001). One study suggested that the number of northern fur seals (*Callorhinus ursinus*) actually entangled were estimated to be 35 times greater than the observed rate at breeding colonies, because entangled animals spend more time at sea foraging and ultimately die from related injuries, infection, or starvation (Fowler et al. 1990).

Reported or observed levels of by-catch should be viewed as minima, because an unknown proportion is likely to go unobserved, thus unreported (Warden and Murray 2011). For example, 83% of Australian sea lions (*Neophoca cinerea*) observed by-caught and drowned in a demersal gill-net fishery in southern Australia actually dropped out of the gear as it was hauled to the surface, before being hauled aboard the vessel (Hamer et al. in press). This provides evidence that additional operational interactions are likely to have occurred, because individuals became by-caught and then either drowned and dropped out at depth, or escaped with life threatening entanglements. Therefore, by-catch of marine mammals is likely to be more extensive than can be revealed through observer programs or fishery logbooks alone.

The life history characteristics of marine mammals indicate they are poorly adapted to rapidly changing conditions (Reynolds et al. 2002). Marine mammals are ‘K strategists’, maintaining their populations around a maximum carrying capacity determined by the finite availability of important resources in the environment, especially food or breeding space (Krebs 1985). This strategy has resulted in marine mammals extending periods of maternal investment to improve survival of their young, which has necessarily resulted in delayed sexual maturity, long life span and low fecundity (Reynolds et al. 2002). For example, of the odontocetes, female common dolphins reach sexual maturity at 2 to 7 years of age, give birth once every 1 to 3 years to a single calf and live up to about 22 years (Perrin 2002). Similarly, female short-finned pilot whales (*Globicephala macrorhynchus*) reach sexual maturity at about 9 years, with birthing intervals of at least 3 years and life expectancy of 60 years (Olson and Reilly 2002). Of the otariids, female Australian sea lions reach sexual maturity at 3.8 to 6.1 years, give birth to a single pup about every 1.5 years and live for up to 24 years (McIntosh 2007).

Many marine mammal species often exhibit population genetic structure (or population units) that is evident at scales ranging from local (e.g. Australian sea lion, Lowther et al. 2012), regional (e.g. bottlenose dolphin, Caballero et al. 2012), or ocean wide (e.g. common dolphin, Bilgmann et al. 2008). This is mainly due to the exhibition of ‘philopatry’, where individual females consistently return to breed at their own place or region of birth, resulting over time in genetic drift caused by geographic isolation (e.g. bottlenose dolphin, McHugh et al. 2011; Australian sea lion, Lowther et al. 2012). Consequently, marine mammal species are often fragmented into genetically distinct populations that in some cases can be small in number. As such, these smaller populations are susceptible to decline and possibly even extinction due to fishery by-catch, because they have limited numbers of sexually mature females that can respond to or compensate for increases in death rates and the associated losses (Moritz 1994).

Depredation by marine mammals is likely to be as widespread as by-catch, with some marine mammal species interacting with all significant fisheries in their range (e.g. California sea lions *Zalophus californianus*; Weise and Harvey 2005). Even though depredating marine mammals risk becoming by-caught and drowned, or acquiring a life threatening entanglement, the individual may gain significant benefits by consuming prey fish caught by the fishing gear, because it negates the need for energetically expensive dives or pursuits, or because they gain access to energy rich fish species that are not a part of their natural diet (Guinet et al. 2007; Hamer et al. 2012).

Depredation can result in a significant cost to the fishery involved, due to the damage or loss of targeted or caught fish. In a Southern Ocean demersal longline fishery targeting Patagonian toothfish (*Dissostichus eleginoides*), killer whale and sperm whale (*Physeter macrocephalus*) depredation resulted in an estimated US$6,052 to US$8,495 loss to the fishery on a daily basis due to damaged product, (calculated in Hamer et al. 2012 from Rouche et al. 2007 and Tixier et al. 2009 taking inflation and exchange rates into account). In the Californian hook and line fishery for Chinook salmon (*Oncorhynchus tshawytscha*), California sea lion depredation resulted in an estimated US$912 to US$2,010 loss to the fishery on a daily basis, amounting to the loss of 8.5% to 28.6% of the catch (calculated from Weise and Harvey 2005 taking inflation into account). Fisheries that experience depredation have 0.6% to 100% of the catch depredated in 2% to 93% of fishing events (e.g. Secchi and Vaske 1998; Hucke-Gaete et al. 2004; Weise and Harvey 2005; Rafferty et al. 2012), thus highlighting the potentially significant and negative impact on commercial fishery operations.

The impact of depredating marine mammals on commercial catch rates is likely to be underestimated, because caught fish can be completely removed and thus not recorded as depredation, or because free swimming fish that might otherwise have become caught are deterred (Read 2005). Gear damage may also occur when marine mammals depredate from fishing gear, with depredating California sea lions causing and estimated US$26,565 to US$89,236 damage to fishing gear across the Californian Chinook salmon fishery (calculated from Weise and Harvey 2005 taking inflation into account). Therefore, costs to a fishery associated with depredation by marine mammals are difficult to quantify, although may include both gear and fish damage in some cases.

Quantitatively determining levels of by-catch and depredation is typically dependent on information collected on the fishing vessels involved. The information can be sourced either from independent observer programs, where a fishery management agency places personnel on vessels to monitor and record information about specific aspects of the fishing operation, or from fishery logbooks, where fishers provide information, either under obligations to licence conditions, or voluntarily (Gilman 2011). Independent observer programs are deemed to be more reliable, because fishers tend to under-report by-catch of and depredation by marine mammals. This is because fishers may be fearful of the repercussions of providing information about by-catch, or may fail to see the value in providing information about depredation (Lewison et al. 2004; Bastardie et al. 2010; Roman et al. 2011). Nonetheless, despite varying degrees of incompleteness or bias, fishery logbooks may provide useful insights into when and where fishing activities occur and provide indicative information about depredation and by-catch. Therefore, they should not be underestimated as a potential source of valuable information for guiding programs designed to mitigate marine mammal depredation and by-catch in fisheries (Gilman et al. 2011).

## 1.3 Past & present

### *1.3.1 Mutual evolution of fish consumption*

The occurrence of interactions between marine mammals and fisheries has doubtless occurred from the first occasion the two sought fish in the same area, many centuries ago. Marine mammals have a long history of occupying the oceanic realm, likely arising from more than one land ancestor between 2 and 27 million years ago (Heyning and Lento 2002). Today, many of the 33 extant species pinnipeds and 72 extant odontocetes eat fish, to a smaller or larger extent, in coastal and oceanic environments, in deep and shallow waters (Riedman 1990; Culik 2004). Therefore, given the intensity of commercial fishing operations today, the potential for one to encounter the other remains high.

Humans have a very short history of harvesting or extracting fish from the oceans by comparison, with sporadic or seasonal coastal harvesting dating back 40,000 years, regular use of nets commencing about 3,000 years ago and distant water pelagic fishing commencing 400 years ago (Sahrhage and Lundbeck 1992). Industrial scale factory fishing (i.e. involving large vessels that can remain at sea for extended periods, processing and freezing large quantities of fish) commenced a mere 50 years ago and resulted in a rapid and global increase in fishing effort. Catch peaked at about 86 million tonnes in 1997, due to discovery and utilisation of previously unobtainable fish stocks (Sahrhage and Lundbeck 1992; Clover 2008; FAO 2009). Exploitation levels have remained steady since then, due mainly to increased fishing effort and capacity, rather than to sustained levels of individual fish stocks (Pauly et al. 2003; Clover 2008; FAO 2009). The need to provide protein to the world's burgeoning human population has increased our interest, and need, to harvest fish. Since the 1970s, advances in industrial design (e.g. more efficient manufacturing processes, stronger materials, high volume hydraulics and more efficiently designed nets) and computer technology (e.g. the introduction of radar, sonar and global positioning system GPS navigation) have facilitated this need, allowing fishing effort to extend further offshore and into deeper waters (Sahrhage and Lundbeck 1992; Clover 2008). Despite the differing longevity of fish exploitation, humans now surpass marine mammals in terms of fish consumption and marine exploitation (Morisette et al. 2010). Therefore, now more than in any other time in history, marine mammal and commercial fishing activities are likely to encounter each other and be involved in operational interactions.

### *1.3.2 Sealing & whaling: exploitation, regulation & international protection*

Marine mammals have been hunted in a subsistence manner for food and clothing since antiquity. The advent of the industrial revolution during the 1700s and 1800s stimulated particular interest in large scale harvesting for marine mammal skin, bones and blubber for the manufacture of clothing, cosmetics, lubricants and lighting fuel. This resulted in industrial scale, numerically intense and geographically widespread exploitation of pinnipeds and cetaceans (e.g. Hiller 1986; Sanderson 1993; Ellis 1999; Ling 1999; Miller et al. 2010).

In Australian territories, the harvest of 180,000 to 193,000 fur seals (possibly Antarctic fur seals *Arctocephalus gazelle*, or subantarctic fur seals *Arctocephalus tropicalis*, or both) occurred on subantarctic Macquarie Island over 10 years from 1810 (Shaughnessy and Fletcher 1987), while approximately 301,080 (including New Zealand fur seals *Arctocephalus forsteri* and Australian fur seals *Arctocephalus pusillus doriferus*) were harvested along the southern Australian coastline over 60 years from 1798 (Warneke 1982; Ling 1999). In the North Atlantic, between 451,000 and 546,000 seals (mainly harp seals *Pagophilus groenlandicus* and hooded seals *Cystophora cristata*) were harvested annually from pelagic exploitation on the Grand Banks over a 20 year period from 1830 (Hiller 1986).

Industrial scale pelagic whaling in Europe and the United States of America (USA) initially focused on the slower baleen whales or mysticetes (e.g. right whales *Eubalaena spp.* and bowhead whales *Balaena mysticetus*) and the only widely targeted odontocete, the sperm whale (*Physeter macrocephalus*), using hand propelled harpoons from open boats (Du Pasquier 1984; Sanderson 1993; Davis et al. 1997; Ellis 1999; Mawar 2000; Miller et al. 2010). Effort in the USA east coast centres of Nantucket and New Bedford alone peaked in 1848 with 10,000 men and 746 vessels (Currie 2001; Boncheva 2011). During the mid 1800s, the depletion of slower species, the development of the rocket propelled 'bomb lance', then the advent of steam and then diesel power, shifted the focus to the more abundant, faster and previously uncatchable *rorquals* (e.g. blue whales *Balaenoptera musculus*, fin whales *B. physalus* and humpback whales *Megaptera novaeangliae*), mostly at higher latitudes (Tonnessen and Johnsen 1982; Ellis, 1999; Clapham and Baker 2002). Over two million were killed in the Southern Ocean alone over 60 years from the mid 1920s (IWC 1995; Yablokov et al. 1998), although several species of odontocete (e.g. sperm whale, pilot whale, killer whale, and at least four dolphin species) were also regularly hunted at possibly unsustainable levels (Clapham and Baker 2002).

Wholesale and unregulated exploitation resulted in the decimation and near extinction of many pinniped populations by the mid 1800s and many cetacean populations by the mid 1900s (Bonner 1982; Hofman and Bonner 1985). Growing awareness that remaining stocks might also go commercially extinct if harvesting restrictions were not applied prompted the development of the *North Pacific Fur Seal Convention 1911*, which banned all pelagic harvesting of northern fur seals (*Callorhinus ursinus*) in all areas under USA, Canadian, Japanese and Russian fishing jurisdiction (Birnie 1986). Similar instruments followed in other regions, including the *North West Atlantic Seals Agreement 1971* (for harp seals and hooded seals; Johnston et al. 2000; Hammill and Stensen 2010) and the *Convention for the Conservation of Antarctic Seals 1972* (CCAS; for elephant seals *Mirounga leonina*, Ross seals *Ommatophoca rossii* and fur seals *Arctocephalus* spp.; Cohen 2002). For cetaceans, harvest regulation has been more unified, in principal (although reality has proven very different, with opposing views on the need for the current whaling moratorium and on the use of whaling for scientific research), with the proclamation of the *International Convention for the Regulation of Whaling 1946* (ICRW), encompassing all cetacean species of commercial harvesting interest (IWC 1946; Birnie 1986).

The 1960s and 1970s heralded an era of growing awareness and concern for the plight of marine mammals (Gordon 1977; Forestell 2002). Specifically, human attitudes evolved from fear and the desire to subjugate and exploit, to compassion, empathy and the desire to conserve (Bearzi et al. 2010; Zelko 2012). The first international effort to define and identify 'endangered' species in need of conservation attention, both marine and terrestrial, occurred with the establishment of the *International Union for the Conservation of Nature* (IUCN) in 1948 (Christoffersen 1997). Among the approximately 105 extant pinnipeds and odontocetes, 24 (about one quarter) are currently included on the IUCN *Redlist of Threatened Species* (Davidson et al. 2012).

Specifically the Redlist classifies seven as *Near Threatened* (five odontocetes and two pinnipeds), eight as *Vulnerable* (five odontocetes and three pinnipeds), six as *Endangered* (one odontocete and five pinnipeds) and three as *Critically Endangered* (one odontocete and two pinnipeds). For many of them, the accompanying listing advice cited commercial hunting as a past threat that may have influenced contemporary population range and size, while for all, commercial fishing was cited as a major contemporary threat to their conservation status.

The IUCN underpinned the development of two important international conservation instruments that have attracted many signatory members desirous of upholding their values, being the *Convention on the International Trade in Endangered Species of Wild Fauna and Flora* (CITES) in 1975, and the *Conservation of Migratory Species of Wild Animals* (CMS or Bonn Convention) 1983. Given that a moratorium on commercial whale hunting under ICRW has been in place since 1986, it also currently acts as a conservation instrument, indicating the changing views of most its members since its inception (Donovan 1992). A similar situation now exists for Antarctic seal species under the CCAS (Peter Shaughnessy, personal communication). In summary, with the exception of a few countries that maintain harvesting activities (e.g. harp seal hunting in Canada and whale hunting by Japan in the Pacific and Southern Oceans), a new era has emerged where conservation of marine mammals is now a strong force in the policy direction of most westernised countries (Forestell 2002; Bache 2003; Gilman 2011).

### *1.3.3 The situation today: the changing seascape & domestic legislation*

Since the cessation of widespread and industrial scale targeting of predominantly baleen whales in 1986 (Clapham and Baker 1999; Miller et al. 2010), the pressure of exploitation has shifted to opportunistic hunting of small odontocetes in coastal waters (Robards and Reeves 2011). Additionally, although widespread seal hunting of pinnipeds has ceased, large quantities of cape fur seals and harp seals are still harvested, with the effects being an ongoing topic of debate (e.g. Stewardson 1999; Leaper et al. 2010). Nonetheless, most marine mammal species now face additional pressure in the form of operational interactions with fisheries. Therefore, it is now likely that some species or populations are under pressure from both intentional and incidental interactions with humans.

The notable recovery of many exploited fur seal populations commenced between the 1950s and 1980s, some 130 to 160 years after the cessation of industrial scale exploitation (e.g. Kirkwood et al. 2010). This delay is attributable mainly to density dependent depression in breeding success (e.g. Roux 1987; Harrando-Perez 2012), a factor that may currently be stymieing the recovery of many odontocete species, which have had much less time to enjoy the euphoria of hunting regulation and protection.

Regardless of whether populations are in an early or advanced stage of recovery from hunting, their population status may be adversely affected by an increased incidence of operational interactions due to the relatively recent and marked increase in fishing effort globally. Due to life history characteristics that typify large mammals, many marine mammal populations may be susceptible to decline if they sustain even small numbers of mortalities that are additional to those occurring naturally, such as those attributable to by-catch. Smaller populations, particularly those in early stages of recovery from hunting, or those being suppressed by the effects of by-catch, may be at greater risk of extinction due to density dependent effects on breeding success (i.e. 'Allee effect'), stochastic variation in size and the occurrence of unpredictable and deleterious events (Gilpin and Soule 1986; Lande 2002; Herrando-Perez 2012). In simplistic terms, these populations cannot afford to lose more than a few individuals to by-catch in a short time period, because they are small and unable to produce offspring rapidly enough to replace those lost, thus may go extinct rapidly.

As populations recover, their increasing numbers concurrently increase the probability of interactions with fishing activities within the overlapping range, because they share finite fish resources (Read 2008). In such cases, full recovery to pre hunting levels may not be possible, because the intrinsic capacity of the population to increase may be neutralised by additional deaths attributable to by-catch. A good example of this involves two dolphin species in the eastern tropical Pacific Ocean. Their numbers were initially depleted by intentional targeting of purse-seiners pursuing tunas (due to an unusual phenomenon known as 'association', where the presence of dolphins indicates the presence of tunas) and the significant by-catch that resulted (Gosliner 1999). Despite the widespread banning of this practice, population recovery appears to have been prevented, at least in part (with broader although likely indirect ecosystem effects such as a documented long-term decline in diatom biomass also possibly responsible; Watters et al. 2003), by continued lower level by-catch, which results in the overall death rate being equal to or exceeding birth rates in their now small populations (Gerrodette and Forcada 2005).

In acknowledgement of these past and present pressures on marine mammal populations, and of obligations pursuant to several international conservation instruments, several nations have developed domestic conservation legislation to combat the problem of incidental marine mammal by-catch in the commercial fisheries they are responsible for (Bache 2003; Gilman 2011). The USA and Australia provide two good examples.

In the USA, the *Marine Mammal Protection Act 1972* (MMPA) embodied the first national approach to conserving marine mammals, by prohibiting the take (e.g. hunting, killing or harassment), import, export and sale of any marine mammal, as a whole, part or product. The development of the MMPA was based on the USA Congressional finding that some marine mammals were at risk of being affected by human activities, specifically citing hunting and fishing activities as threats. It was deemed that this finding justified the need to encourage research and conservation activities and discourage activities that would result in further threats to populations that could result in numeric decline (MMC and NMFS 2007).

In Australia, the states provide varying protection for marine mammals. For example, in Tasmania (Australia), New Zealand fur seals and Australian fur seals have been partially protected since 1891 and fully protected pursuant to current state legislation since 1970 (MMIC 2002). A similar situation exists in Western Australia and South Australia, where dedicated legislative protection for both fur seal species and the Australian sea lion has been in place since 1950 and 1964, respectively (Warneke 1982; NSSG and Stewardson 2007). However, state jurisdiction only extends from the coastline to 3 nautical miles offshore, with the Australian (Commonwealth) Government responsible for waters from that point out to 200 nautical miles offshore (AGD, 1980). Despite efforts in each state, the Australian Government was slow to provide statutory protection to marine mammals. Seals and dugongs were first protected in 1987 from deliberate interference and neglect pursuant to the now rescinded *National Parks and Wildlife Act 1975*, which lacked a process to address human impacts, including fisheries by-catch (Bache and Evans 1999). A significant improvement occurred with the subsequent proclamation of the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act), under which it is now an offence to kill, injure, take, trade, keep, or move species that are listed, threatened, or migratory, which includes all pinnipeds and cetaceans, in its area of jurisdiction (DSEWPaC 2001a, 2001b).

The Department of Sustainability, Environment, Water, Population and Communities (DSEWPaC), as administrator of the EPBC Act, also have provisions for assessing the impact of commercial fisheries on marine mammal populations (and any other native marine population or community). In practice, commercial fisheries operating in Australian territorial waters require approval from the DSEWPaC to conduct a wildlife trade operation (WTO) for an exempted native species (i.e. a wild fish species). The process is underpinned by the *Guidelines for the Ecologically Sustainable Management of Fisheries*, with one of its principals being that "*Fishing operations should be managed to minimise their impact on the structure, productivity, function and biological diversity of the ecosystem*" (DEWR 2007). Generally every five years, each commercial fishery must undergo an *environmental assessment* by its manager pursuant to the guidelines, and pursuant to specific requirements in Part 10 (strategic assessments) and Part 13 (native species and communities) of the EPBC Act (DEWR 2007). The DSEWPaC, after consideration of the environmental assessment, will typically respond with a suite of conditions aimed at improving the activities of the fishery in a way that will reduce its impact, which must be actioned by the fishery or fishery manager within prescribed timeframes. In Australia, these conditions often provide the impetus for initiating studies or actions aimed at mitigating the often deleterious impact of commercial fisheries on marine mammals, which applies to the case studies presented in this thesis, particularly those in chapters 4 and 6.

### 1.4 Mitigation strategies

In reality, attempts to mitigate by-catch and depredation are a luxury enjoyed almost exclusively by developed or western countries, because they have the necessary financial prosperity, research institutions and governance structures to develop, test and implement them (Gordon 1977). Nonetheless, implementing change in response to reports of operational interactions have often been delayed, sometimes indefinitely, because opinions on the best approach to mitigating by-catch and depredation are seldom unified, due to the differing goals of the exploiters and the conservers (Bache and Evans 1999). This is despite the longstanding recognition that marine mammal by-catch occurs in many fisheries carried on by developed countries around the world (e.g. Hall et al. 2000). Even in situations where legislation is developed to protect marine mammals, limited capacity or will to monitor or enforce compliance can sometime result in little or no change (e.g. Mangel et al. 2010). In contrast, developing nations are typically ill equipped to fund, monitor or enforce the implementation such initiatives. This outcome is primarily due to a general lack of funds to drive education and capacity, although it is exacerbated by the fact that many poorer fishers in developing take advantage of the incidental by-catch of some marine mammals species as a food source, motivated either by cultural frameworks or necessity (Mangel et al. 2010; Robards and Reeves 2011).

As chapter 2 will reveal, attempts to mitigate by-catch and depredation of odontocetes in pelagic longline fisheries is relatively recent. A similar situation exists for most instances around the globe where marine mammals have operational interactions with commercial fisheries. When attempting to mitigate marine mammal depredation and by-catch at the vessel, three broad approaches can be identified, being modifications to (i) fisher behaviour, (ii) fishing gear and (iii) fishing effort distribution. These three approaches are the focus of this thesis.

There are other approaches to mitigation that have been discussed in the published literature, being the use of acoustic deterrence and the effect it has on the behaviour of marine mammals. Neither of these two aspects will be explored further here, due to three key limitations associated with the current technology and electronic devices it depends on as largely impractical or inappropriate. Specifically, these limitations are due to (i) typically large transponders and batteries (Mooney et al. 2009), (ii) in some cases rapid sound attenuation that markedly reduces effectiveness with distance and (iii) in other cases excessive noise production that may have unintended although adverse impacts on the broader marine environment (Morton and Symonds 2002), and (iii) a limited understanding of the mechanism of deterrence that has varying effectiveness on different species, which in some cases diminishes over time or has the opposite effect (Jefferson and Curry 1996). A more detailed discussion of these limitations and of acoustic deterrence in general is available in chapter 2.

This thesis focuses on industrial scale commercial fisheries in the Oceania region. This is mainly because there is a prevalence of examples of operational interactions between marine mammals and fisheries in the region that need to be addressed for conservation and economic reasons, and because there is access to sufficient financial and legislative support to identify, develop and test strategies for eventual implementation.

### *1.4.1 Improved fishing behaviour*

Improving fisher behaviour is perhaps the most immediate and cost effective method of for mitigating by-catch and depredation in theory, because it simply requires consensus between stakeholders in order for implementation to become possible. Trawl and purse-seine fisheries lend themselves well to this approach.

The by-catch of Australian fur seals prompted a Tasmanian freezer trawler fishery, targeting the large benthopelagic blue grenadier (*Macruronus novaezelandiae*), to modify its fishing practices during the early 2000s, using a fishery Code of Practice (CoP). Individuals attempting to depredate the caught fish enter the front of the net and then are unable to escape, often due to the forward speed of the net through the water. Pursuant to the CoP, vessels were required to (i) steam at 10-12 knots for 40 minutes prior to setting gear if animals were seen immediately before setting the gear, (ii) never raise the gear to shallower than 150 m during a fishing event and (iii) remove all fish stuck in the net after each haul that might serve as an attractant during the next fishing event (SETFIA 2000; Hamer et al. 2006; Tilzey et al. 2006). Although a seemingly simple approach, these actions were likely come at some cost to the fishing operation, suggesting that compliance to the CoP would likely be minimal without observers monitoring fishing activities. Nonetheless, it was difficult to determine the specific success of the CoP in isolation, although its use alongside gear modifications (i.e. the inclusion of a seal exclusion device, or SED, in the cod-end of the trawl net) resulted in a 50% reduction in by-catch (Tilzey et al. 2006). Despite the occurrence of fur seal by-catch in the blue grenadier fishery and the fact that Australian fur seals target small epipelagic fish, such as redbait (*Emmelichthys nitidus*) and blue mackerel (*Scomber australasicus*; e.g. Deagle et al. 2009), a similar CoP was not trialled in a small pelagic trawl fishery operating in the same area until several years later (Lyle and Willcox

2008). This situation highlights the need for greater cross communication between fisheries and fishery managers about marine mammal by-catch issues.

The intentional encirclement of spotted and spinner dolphins in purse-seine nets eastern tropical Pacific Ocean (ETP) to catch associating tunas resulted in large numbers being killed, averaging 378,632 ± 123924 animals during the 1960s (Gosliner 1999). This untenable situation initially prompted the introduction of the 'back down' method in the US fleet, where the vessel was put astern and the float line was dragged below the surface to facilitate the escape of encircled animals (Gosliner 1999). Additionally, it was later decided that vessels should refrain from targeting dolphins in this way altogether. These improved practices were attributable for the ~95% reduction dolphin mortalities, averaging 16,984 ± 4,369 animals during the 1980s (Gosliner 1999). However, it was acknowledged that some incidental by-catch was likely to continue and that intentional encirclement continued in the non US fleet (which had expanded markedly during the 1970s and 1980s), due to limited adoption of the two improved fishing practices.

The strategy of improving fishing practices is explored in chapter 4, where a CoP is implemented to mitigate dolphin by-catch in a purse-seine fishery in South Australia (SA). As will be explained, it was found that key changes to fisher behaviour, which had little impact on the fishing operation overall, could markedly reduce dolphin by-catch.

### *1.4.2 Modified fishing gear*

In situations where improvements to fisher behaviour to address the problem are insufficient, not practical or relevant, or where a consensus cannot be reached, gear modification may be a viable alternative in some fisheries. In the case of the ETP dolphin by-catch problem, the use of improved fishing practices alone was deemed insufficient to reduce mortalities to acceptable levels. As such, the purse-seine net was modified to include a ‘Medina panel’ and a ‘porpoise apron’, both of which were designed to ‘stiffen’ the walls of the net and thus reduce the chance that dolphins would become physically tangled and drown (Coe et al. 1984).

Similarly, the Tasmanian blue grenadier and New Zealand squid trawl fisheries have implemented fishing gear modifications, because the use of CoPs alone have been unable to prevent the drowning of Australian fur seals and of New Zealand sea lions (respectively). It is thought this is largely because individuals enter the front of the net to depredate caught fish and then are unable to escape due to the forward speed of the net through the water and due to some level of disorientation (DJH; unpublished video footage). Both fisheries have sought to address this problem and mitigate drowning events by placing a rigid, stainless steel ‘grid’ and a modified net ‘escape hatch’ just ahead of the cod-end, where caught fish are collected. The grid is designed to prevent fur seals from being smothered by caught fish as they were funnelled into the cod-end and the escape hatch was designed to allow those fur seals inside in net to exit immediately, without having to swim against the current inside the large trawl net structure (Wilkinson et al. 2003; Hamer et al. 2006). However, most studies of the effectiveness of the grid and escape hatch have been marred by an inability to separate the effect of CoPs and by a lack of standardisation of configuration, which is highlighted in one study where several configurations were used haphazardly (Tilzey et al. 2006). Therefore, future efforts to determine the effect of gear modifications designed to mitigate marine mammal by-catch in trawl fisheries should be guided by sound experimental principals that allow the benefit of specific gear modifications to be quantified.

These examples illustrate the potentially wide scope to modify fishing gear to mitigate by-catch, although two examples highlight that other fisheries may also be motivated to implement gear modifications to prevent catch depredation. Firstly, a Chilean demersal longline fishery has trailed the use of 'net sleeves' on branchlines (Moreno et al. 2008). The device is designed to remain clear of the baited hook and then descend under the weight of gravity to shroud any caught fish when the gear was hauled. Their use has substantially reduced depredation by killer whales and sperm whales in that fishery (Moreno et al. 2008). Secondly, southern Australian rock lobster pot fisheries have trialled or implemented the use of 'spikes' (Campbell et al. 2008; Goldsworthy et al. 2010). The spike, a straight and vertical rod of about 20 mm in diameter that attaches to the base of the pot and protrudes into the entrance some 200 mm above, thus allowing rock lobsters to enter while simultaneously preventing larger seals from entering and depredating rock lobsters inside the pot. Although only preliminary experimental trials have been undertaken to date, depredating sea lions were found to be less successful at depredating rock lobsters from pots when the spike was in place (Hamer et al. in prep.).

These examples demonstrate that gear modifications can be used to benefit marine mammals by mitigating by-catch and commercial fisheries by mitigating depredation. Mitigation of by-catch may be a benefit of mitigating depredation, because preventing depredating marine mammals from coming into contact with fishing gear abolishes the chance that they might become by-caught. Chapter 3 explores this two-pronged benefit and finds that, while more data needs to be collected to confirm the effectiveness of developmental devices in deterring depredating odontocetes, they have little impact on other aspects of the fishing operation.

#### *1.4.3 Reduced spatial overlap*

When changing fisher behaviour or modifying fishing gear is impractical or unsuccessful in achieving management objectives to mitigate operational interactions, which may be either to mitigate or abolish by-catch, then reducing the degree of spatial overlap between the two marine predators may be the only viable alternative. This method is generally viewed as a last resort, because it generally involves limiting where and when a fishery can operate and thus is likely to have a negative economic impact on the fishery involved (Bache 2003).

There are two methods used for implementing spatial closures. The first method requires the statutory fishery management agency to impose conditions on a fishing licence that prohibit activities in specified areas, either permanently, at specific times (e.g. when a marine mammal species migrates through an area; Hyrenbach et al. 2000), or under certain conditions (e.g. when a by-catch limit is reached; AFMA 2012). In the case of fisheries operating in Australian waters, this would be the Australian Fisheries Management Authority (AFMA). Chapter 6 explores the level of overlap and by-catch of the Australian sea lion with a demersal gill-net fishery. It concludes that while AFMA has taken considerable steps using permanent and seasonal closures to mitigate the impact of operational interactions on Australian sea lion populations, further efforts may be necessary to reduce levels of by-catch to acceptable levels.

The second method and the associated outcome is identical in practice, with the main difference being that it is externally imposed by a statutory conservation management agency as a marine protected area (MPA), rather than by the fishery management agency. For fisheries operating in Australian waters, a situation such as this might occur if DSEWPaC proclaimed an MPA in an area utilised by an AFMA managed fishery. For example, in Mexico it was determined that an incremental increase in the size of an MPA across the area used by the vaquita (*Phocoena sinus*; a small and endangered coastal porpoise) would reduce its overlap with a gill-net fishery in which individuals of the populaitons frequently became by-catch (Gerrodette and Rojas-Bracho 2011). This approach seems to have improved the conservation outlook of the population. Similarly, chapter 5 attempts to assess the effectiveness of an MPA intended to protect a small and isolated population of Australian sea lions from by-catch in demersal gill-nets. The study concluded that greater restrictions to fishing were needed within the MPA at times when it was open to fishing and that it needed to be much larger generally, in order to improve the conservation protecton of resident Australian sea lions.

## 1.5 Need & aims

Interactions between marine mammals and fisheries are on the increase globally, due either to a greater level overlap between the two, due in part to increased fishing capacity and effort, or to an increased desire or capacity to report, due to an increased awareness and desire to manage fisheries sustainably. Trophic competition is cryptic, complex and difficult to characterise, thus development of appropriate and effective solutions has proven challenging. In contrast, operational interactions (of which marine mammal depredation and by-catch are the main concerns) are conspicuous and comparatively easy to characterise. Despite this situation seeming to lend itself to straightforward characterisation of the problem and of development and assessment of mitigation strategies, there have been surprisingly few documented attempts to do so.

The paucity of attempts may be attributable to two factors. Firstly, there is a general lack of consensus within and between statutory, fishing and scientific communities about how best to identify and tackle associated problems, due mainly to differing values and thus motivation among the stakeholders (Bache 2003; Gilman 2011). Secondly, developing and assessing potential solutions often requires a considerable commitment of time and funds, both of which are typically in short supply under contemporary conventional funding models (e.g. West et al. 1999; Lowry et al. 2011). Nonetheless, there is a growing need, at a local as well as a global scale, to develop and assess strategies to mitigate operational interactions between marine mammals and fisheries, with a view to assist in the ecologically sustainable development of fisheries and in the conservation of marine mammals. In general, this thesis aimed to make significant inroads into resolving a number of marine mammal by-catch and depredation issues identified in the Oceania region, specifically in SA, the Coral Sea and in waters around Fiji. More specifically, this thesis aimed to:

1. Review a major fishing method in the two regions in which there are operational interactions with marine mammals, where both by-catch and depredation occurs and which is likely to become an increasing problem on a global scale;
2. Characterise the nature and extent of depredation and by-catch where operational interactions are known to exist between marine mammals and commercial fisheries in the Oceania region; and
3. Where deemed necessary in those fisheries, develop mitigation strategies and explore their efficacy.

### 1.6 Focus & structure

Each of the five research chapters in this thesis are stand alone case studies of operational interactions between marine mammals and commercial fisheries. Chapter 2 provides a general review and characterisation of the problem of depredation and of by-catch in longline fisheries and introduces the idea of modifying the fishing gear to mitigate both problems. These two problems have emerged as a concern internationally, with many longline fisheries reporting significant economic impacts caused by depredation and many conservation groups increasingly alarmed about the impact of by-catch on many odontocete populations. Chapter 3 builds on this theme by exploring in more detail how two devices, currently being developed and trialled in Australian and Fijian pelagic longline fisheries, might be able to mitigate depredation and by-catch simultaneously, using physical and psychological deterrence. Chapter 4 attempts to address DSEWPaC conditions to conserve dolphin populations that interact with a purse-seine fishery in SA, by characterising the problem and quantifying by-catch and encirclement rates, and then documenting attempts to mitigate the problem by modifying fishing gear and fisher behaviour. Chapters 5 and 6 also attempt to address DSEWPaC conditions to conserve the Endangered Australian sea lion, which becomes by-catch in a demersal gill-net fishery operating in SA. Specifically, Chapter 5 assesses the performance of an MPA (the Great Australian Bight Marine Park, or GABMP) in protecting a small population residing in the area and Chapter 6 quantifies the extent of overlap and level of by-catch between the fishery and all populations residing in South Australia as a means informing the relevant statutory management agencies.

All five research chapters have been published; four in peer reviewed scientific journals and one as a peer reviewed Australian Government report. As such, there are some differences between the style and layout and the more traditional thesis format. Firstly, there is some repetition between chapters in the explanation of some underlying concepts, methods and general descriptions, and each chapter has a stand alone reference section. Similarly, acronyms and binomial genus and species names are included with the first mention of a term or species, respectively, in each chapter. Secondly, there are journal specific editorial variations between chapters, including variations in the presentation of words (e.g. by-catch or bycatch, 'operational' or *operational*, demersal or benthic), units (e.g. month or mo), figures and tables (mainly in their visual layout).

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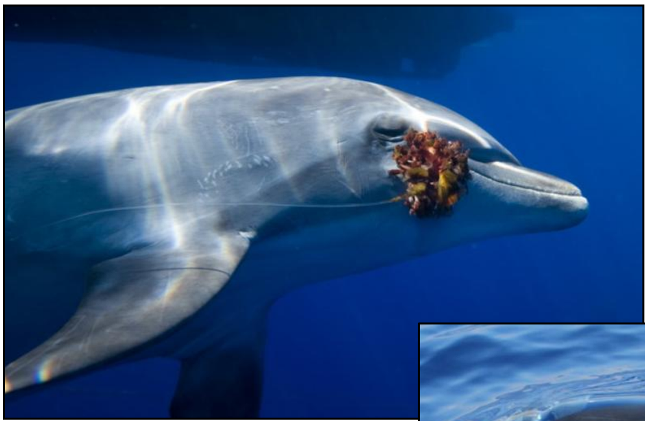
## Odontocete bycatch and depredation in longline fisheries: a review of available literature and of potential solutions



*It's better to look at the sky than live there. Such an empty place; so vague. Just a country where the thunder goes and things disappear.*

Truman Capote

## Statement of Authorship

Title of publication: Odontocete bycatch and depredation in longline fisheries: a review of the problem and of potential solutions

Publication status: Published, February 2012

Publication details: *Marine Mammal Science* 28, E345-E374

### Author Contributions

By signing the Statement of Authorship, each author certifies that their stated contribution to the publication is accurate and that permission is granted for the publication to be included in the candidate's thesis.

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## 2.1 ABSTRACT

Operational interactions between odontocetes (*i.e.*, toothed whales) and longline gear are a global phenomenon that may threaten the conservation of odontocete populations and the economic viability of longline fisheries. This review attempts to define the issue, summarize the trends and geographical extent of its occurrence over the last half century, explore the potential impact on odontocetes and on fisheries, and describe potential acoustic and physical mitigation solutions.

Reports of odontocete bycatch rates are highly variable (between 0.002 and 0.231 individuals killed per set; the number of hooks per set is rarely recorded In logbooks) and at least 20 species may be involved. Information about marine mammal population size, migration patterns and life history characteristics are scarce, although at least one population may be in decline due to losses attributable to longline bycatch. Information about the financial impact of depredation on pelagic longline fisheries is also scarce, although estimates of daily fleet-wide losses range between US$1,034 and US$8,495 (overall fleet income was not reported). Such biological and financial losses may be unsustainable.

Recent developments in acoustic and physical mitigation technologies have yielded mixed results. Acoustic mitigation technologies have no moving parts, although require complex electronics. To date, they are insufficiently developed and their efficacy has been difficult to assess. Physical mitigation technologies generally require complex moving parts, although they are relatively simple to develop and assess. Further development and testing remains necessary before widespread implementation would be possible. Development of these approaches should be prioritized and a ‘toolbox’ of various strategies and solutions should be compiled, because a single panacea to the problem is unlikely to emerge.

## 2.2 Introduction to operational interactions between odontocetes and longlines

The occurrence of operational interactions between cetaceans (*e.g.*, whales, dolphins, and porpoises) and commercial fisheries has attracted considerable attention in the published literature and is a familiar problem to many fishing, management, and research communities (*e.g.*, Northridge 1984, 1991; Beverton 1985; Reeves *et al.* 1994; Northridge and Hofman 1999; Donoghue *et al.* 2003; Shaughnessy *et al.* 2003; Read 2005; Gilman *et al.* 2006). Operational interactions involve the simultaneous physical convergence of cetaceans and commercial fisheries upon the same spatially retracted area, often when both are in pursuit of the same fish (Northridge and Hofman 1999, Shaughnessy *et al.* 2003, Hamer *et al.* 2008, Moreno *et al.* 2008). Positive outcomes include (1) fisheries using cetaceans to indicate the presence of fish (Gosliner 1999, Northridge and Hofman 1999) and (2) cetaceans using fisheries to access an otherwise inaccessible food resource (Gilman *et al.* 2006, Moreno *et al.* 2008). Negative outcomes include threats to the viability of (1) cetacean populations when depredating individuals become bycatch and are injured or killed (Gosliner 1999, Shaughnessy *et al.* 2003, Hamer *et al.* 2008) and (2) fisheries when depredating whales remove or damage the catch (Hucke-Gaete *et al.* 2004, Ramos-Cartelle and Mejuto 2008). The negatives may be exacerbated by an increase in the level of trophic interactions due to increased competition for the same fish stock, which results in either direct reduction (through removal of fish), or indirect reduction (through trophic cascades) of target fish stocks (Northridge and Hofman 1999, Kaschner 2004). Both scenarios could reduce the overall quantity of fish available to cetacean populations (Kaschner 2004, Bakun *et al.* 2009) and fisheries (Ashford *et al.* 1996, Earle 1996), thus increasing the likelihood of operational interactions.

A growing body of information concerning the nature and extent of operational interactions between odontocetes (*i.e.*, toothed whales, dolphins, and porpoises) and longline gear has been emerging in the literature, since longlining commenced modernization in the 1950s (Yamaguchi 1989, Ward and Hindmarsh 2007). The main areas identified to be of concern are depredation and bycatch. Depredation occurs when an individual odontocete partially or completely consumes caught fish from the longline, or deters free swimming fish that may otherwise have become caught (Yano and Dahlheim 1995, Northridge and Hofman 1999, Read 2005, Gilman *et al.* 2006, Lauriano *et al.* 2009). Although depredation of bait has also been identified as an issue, there is insufficient data and information currently available to warrant in-depth consideration here. Bycatch occurs when a depredating odontocete becomes caught on a longline hook when attempting to remove the catch (Beverton 1985, Shaughnessy *et al.* 2003, Read 2005, Secchi *et al.* 2005, Indian Ocean Tuna Commission 2007).

The modernization of longlining mid-last century resulted in rapid geographic expansion, with reports of odontocetes depredating catch from longlines emerging soon after (*e.g.*, Iwashita *et al.* 1963, Sivasubramaniam 1964, Mitchell 1975). The growing number of reported incidences since that time suggests the phenomenon may have become a significant economic problem for affected longline fisheries and a significant conservation and welfare problem for affected odontocete populations. This review attempts to (1) define the issue, (2) summarize the available literature to determine temporal trends and geographical extent of catch depredation by and bycatch of odontocetes, (3) explore the impacts of depredation on fisheries and bycatch on odontocete populations, and (4) describe the acoustic and physical tools being developed to mitigate the problem.

### 2.3 Summary of odontocete depredation & bycatch reports in the reviewed literature

The compilation and interpretation of the accessible literature may be useful in revealing the nature and extent of the problem and may provide insights into how to mitigate the impact of one or both. The literature cited was restricted to peer reviewed documents (articles and reports) that referred to fishery logbook or observer data, specifically relating to operational interactions between odontocetes and longline fisheries. Electronic search engines and databases were used, such as Web of Science, Current Contents, Google Scholar, and general internet searches, using keywords such as: whale, cetacean, odontocete (and individual species names), depredation, and bycatch.

The literature search identified 32 peer reviewed documents matching the specified criteria, published between 1964 and 2010 (Table 1). Early documents merely acknowledged the occurrence of catch depredation, with the first specific account of an odontocete being bycaught on a longline hook emerging in 1983 (Di Natale and Mangano 1983). Nonetheless, the literature has remained focused on the effects on the fishery, with 23 reports of depredation compared with 12 reports of bycatch (Table 1). Twenty-two reports have emerged since 2000, amounting to over twice the number produced over the previous four decades combined (Fig. 1). This recent spike suggests an increase in awareness and interest in the issue.

The literature cited indicates that 20 odontocete species have been involved in operational interactions with longline gear. Fifteen species were confirmed to have either depredated from, or have become bycaught on, longline hooks (Table 1). The five remaining species (*i.e.*, rough toothed dolphin, *Steno bredanensis*; spinner dolphin, *Stenella longirostris*; Atlantic humpback dolphin, *Sousa teuszii*; melon-headed whale, *Peponocephala electra*; and pygmy killer whale, *Feresia attenuata*) were mentioned in the literature cited, but were involved in unverified, anecdotal and unquantified events (Northridge 1984, Nishida and Tanio 2001, South Pacific Regional Environment Program 2002, Culik 2004, Secchi *et al.* 2005,Watson and Kersletter 2006, Moore *et al.* 2010). Based on the literature obtained, killer whales (*Orcinus orca*) and sperm whales (*Physeter macrocephalus*) appear to be the main species involved with demersal longline fisheries at higher latitudes, while false killer whales (*Pseudorca crassidens*) and pilot whales (*Globicephala* spp.) appear to be the main species involved with pelagic longline fisheries at lower latitudes. The problem also appears to be geographically widespread, with reports of depredation from and bycatch on longlines confirmed in 25 locations, from the equator to high latitudes in both hemispheres and in all of the world's major oceans (Fig. 2).

Some key events may explain the recent increase in the number of reports emerging in the available literature. Since the 1940s and 1950s, some odontocete populations appear to have benefited from increased international protection instruments, such as the Convention on International Trade in Endangered Species (CITES) and the International Convention for the Regulation of Whaling (ICRW). ). These conservation instruments reflect a growing global awareness of the problem, its impact on marine mammal populations and the need to mitigate it. During the same period, fishing effort has increased to meet the demands of a burgeoning human population (United Nations 2009) with changing dietary needs (Duarte *et al.* 2009). This situation is likely to have increased the probability of odontocetes encountering fishing gear, thus resulting in increased incidences of depredation and bycatch (Northridge 1984, 1991; Jefferson 1994; Reeves *et al*. 1994; South Pacific Regional Environment Program 2002; Donoghue *et al.* 2003; Gilman *et al.* 2006). As such, the recent growth in the volume of literature may reflect an increase in the motivation of fishermen to find ways of mitigating catch depredation, in a bid to improve or maintain catch returns, at a time when increased operational costs (*i.e.*, fuel and freight) and depleted fish stocks (*i.e.*, overfishing) are eroding profits (Northridge and Hofman 1999, Ebert *et al.* 2009, Food and Agriculture Organization 2009). The emergence of this information has encouraged relevant conservation and management organizations to characterize the problem and to explore mitigation strategies that facilitate the continued conservation of recovering cetacean populations and to minimize mounting pressure on the economic viability of fisheries. Mitigation of odontocete bycatch and catch depredation has been prioritized by some fisheries in recent times, indicating that stakeholders rank its importance highly, relative to other issues that impact fishery viability (Donoghue *et al.* 2003, Australian Fisheries Management Authority 2005).

### 2.4 Impacts of bycatch on odontocetes

Longline gear poses a significant injury and drowning risk to depredating odontocetes, which affect the welfare of individuals and the conservation of populations (Ashford *et al.* 1996, Northridge and Hofman 1999, Visser 2000, South Pacific Regional Environment Program 2002, Secchi *et al.* 2005, Gilman *et al.* 2006, Forney and Kobayashi 2007, Hamer 2009*a*, Lauriano *et al.* 2009, Reeves *et al.* 2009). Some individuals may accidentally ingest a hook when they depredate catch from longline hooks, which may become lodged in their mouth, throat, or stomach (Secchi *et al.* 2005, Fig. 3A). These events may lead to internal injuries, infections, starvation, or even eventual death (Best *et al.* 2001). Some hooked individuals may be unable to reach the surface to breathe, thus leading to a more immediate death by drowning (Hamer 2009*a*). Depredating odontocetes are also often conspicuous, especially during hauling when they are close to the vessel, which may lead to fishermen becoming frustrated and attempting to shoot individuals (Northridge and Hofman 1999). The impact of these mortalities at a population level is difficult to determine, because there are currently inadequate data available to estimate bycatch or mortality levels, or to determine historical and current levels (thus trends) in population size.

The distributions of most of the 72 extant odontocete species overlap geographically with longline fishing activities in some part of their range (Northridge 1984, Bjordal and Løkkeborg 1996, Culik 2004, Carwardine 2006). This is corroborated by the occurrence of operational interactions of 20 of the 72 extant odontocete species with longline fisheries (Table 1). The literature cited provides insights into this issue, although there is a much larger source from anecdotal, qualitative, unverified, and fishery dependent reports, indicating the occurrence of a much larger and chronic welfare and conservation problem.

A recent and intensive study of operational interactions between a false killer whale population and a pelagic longline fishery in Hawaiian waters (based on independent longline observer programs and odontocete population surveys during the 1990s and 2000s) concluded that the incidence of bycatch increased the risk of population decline (Forney and Kobayashi 2007, Reeves *et al.* 2009). Nonetheless, establishing a robust quantitative link is difficult in the absence of reliable estimates of odontocete bycatch and populations, and of quantitative overlap, which is unlikely to occur with the data currently available (Hamer *et al.* 2008, 2009, Indian Ocean Tuna Commission 2010). This problem is further exacerbated by under reporting in fishery logbooks, which occurs because fishermen are typically fearful of the negative consequences of reporting accurately (Moore *et al.* 2010).

Recent advances in population genetics have made it possible to identify 'management units', which may assist in ensuring the biological importance of subpopulations is not underestimated (Pimper *et al.* 2010). Notwithstanding, most odontocete species have low reproductive rates and correspondingly low intrinsic capacities for increase, suggesting that even low levels of additional or unnatural mortality may cause decline (Leatherwood *et al.* 1983,Wade 2002, Culik 2004, Miller 2007). This is further complicated by the growing number of reports of genetic subdivision within what were previously thought to be single populations (*e.g.*, killer whale: Pilot *et al.* 2010; false killer whale: Chivers *et al.* 2007; common dolphin: Bilgmann *et al.* 2008; bottlenose dolphin: Krutzen *et al.* 2004). Nonetheless, this technology will allow more appropriate management strategies to be developed and implemented in the future, which take into account the genetic diversity *within* a species.

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Although an individual odontocete is faced with the risk of injury or death when depredating from longlines, it may receive considerable foraging and energetic benefits by doing so. Some fish species caught on longlines may be unavailable to depredating odontocetes under natural conditions, because those fish are too large, too fast to catch, or occur in very deep waters (Gilman *et al.* 2006, Tixier *et al.* 2009). Fish caught on longlines may offer an energetic advantage to depredating odontocetes, because they can be consumed without the need for deep dives or prolonged pursuits (Guinet *et al.* 2007). If an individual odontocete can develop a strategy to avoid becoming bycatch and the activities of the longline fishery they depredate from is frequent and predictable, then they may be at an energetic advantage compared with other individuals of the same species that forage naturally.

## 2.5 Impacts of depredation on longline fisheries

Although concerns about the welfare and conservation of depredating odontocetes have become more common in recent times, concerns about the economic impact of depredation on affected longline fisheries have persisted since the 1960s (Dahlheim 1988,Yano and Dahlheim 1995, South Pacific Regional Environment Program 2002, Australian Fisheries Management Authority 2005, Indian Ocean Tuna Commission 2007). Depredation can reduce the overall size and condition of the landed catch, because target fish may be deterred from taking baited hooks, or caught fish may be damaged or removed completely (Yano and Dahlheim 1995, Northridge and Hofman 1999, Gilman *et al.* 2006, Hamer 2009*b*, Lauriano *et al.* 2009). When depredation occurs, affected fisheries are likely to experience sporadic, seasonal, or ongoing reductions in profit, which may lead to economic decline, especially if the problem persists.

When depredating odontocetes attack fish caught on longline hooks, they often remove the entire torso from behind the gill plates (Fig. 3B), or leave tooth lacerations on the torso (Fig. 3C).

Odontocete teeth are pencil-like and tend to tear the skin and flesh of the caught fish, causing extensive damage. The nature of this damage is distinct from that caused by sharks, which tend to remove clean, bite-shaped portions of flesh from the torso of caught fish with their blade-like teeth, leaving the surrounding flesh relatively undamaged (Fig. 3D). Distinguishing between odontocete and shark depredation is important for (1) ensuring the correct attribution of damage to each depredating taxa and (2) selecting the correct mitigation method. Anecdotal accounts suggest that odontocetes may also be blamed for shark depredation. This is because depredating odontocetes surface frequently to breathe often in the vicinity of fishing vessels, this are much more conspicuous than sharks. These instances may result in the overestimation of whale depredation and the underestimation of shark depredation.

In addition to impacting directly on the catch, depredating odontocetes may damage the fishing gear when they remove caught fish, specifically hooks, or larger portions of the longline gear, especially if they become caught themselves (Northridge and Hofman 1999, Gilman *et al.* 2006). Furthermore, small odontocetes (*i.e.*, dolphins) may partly (Fig. 3E) or completely remove baits directly from the hook (Secchi *et al.* 2005, McPherson *et al.* 2008). However, large quantities of small pelagic fish have been observed grazing on discarded baits and around baited hooks as they are hauled aboard the vessel at the end of a set (Hamer, personal observation; Fig. 3F). This suggests small pelagic fish or squids, rather than odontocetes, may be responsible for depredating baits on some occasions.

The damage to caught fish and deterrence of target fish by depredating toothed whales is likely to result in financial losses for affected fisheries. Two studies conducted in a demersal fishery fishing in the Bering Sea (northeast Pacific; for halibut, *Reinhardtius hippoglossoides*, and arrowtooth flounder, *Atheresthes stomias*) between 1977 and 1989 estimated the daily economic cost of killer whale depredation to each vessel was between US$1,034 and US$8,449

(Dahlheim 1988, Yano and Dahlheim 1995). These figures were based on one set per day and 78.9% inflation between 1989 and 2010 for the earlier study and 44.2% between 1995 and 2010 for the later study. Two other studies conducted around the Crozet (46°25'S, 50°59'E) and Kerguelen Islands (49°19'S, 69°28'E), Southern Ocean, between 2003 and 2008 estimated the daily cost of killer whale and sperm whale depredation to the fishery across a French demersal fishery (for Patagonian toothfish, *Dissostichus eleginoides*) was between US$6,052 and US$8,495 (Roche *et al.* 2007, Tixier *et al.* 2009). These figures were adjusted from a multiyear to a one set per day estimate, then 1.5:1€ to US$ exchange conversion and 13.6% inflation between 2005 and 2010 for the earlier study and a 1.3:1€ to US$ exchange conversion and 2.7% inflation between 2008 and 2010 for the later study. It is important to note that these figures are at best informative and represent a snapshot in time for two demersal longline fisheries in the northeast Pacific and the Southern Ocean. They are unlikely to reflect the losses sustained by other demersal or pelagic longline fisheries in other locations at other times. In addition, neither study reported the overall catch figures for vessels or for the fleet, thus it was not possible to determine the percentage of catch that was lost, nor the impact on profits. Furthermore, the economic cost of depredation is likely to be an underestimate, because it is not possible to quantify the number of caught fish that are completely removed from the hook, the number of target fish that are deterred from taking a baited hook altogether, the amount of fishing gear that is damaged, nor the various avoidance activities undertaken by skippers (Yano and Dahlheim 1995, Hamer 2009*b*). Despite the lack of data, the economic costs reported in the two studies detailed suggest that the fishery wide economic impact is likely to be significant.

When depredating whales completely remove caught fish from longline hooks, they may impede effective fishery management practices, such as setting accurate total allowable commercial catch (TACC) limits and calculating the catch per unit of effort (CPUE) for target fish stocks. Overfishing may occur, because the caught fish that are removed by depredating odontocetes are not included when calculating the level of exploitation of the targeted fish (Gilman *et al.* 2006). This situation may occur when the TACC has already been set, because the depredated fish are not included in the catch declared by the fishery. Under fishing may occur, because the removal of caught fish by depredating whales will reduce the CPUE. This situation may lead to the false impression that there are less fish available than is actually the case, which may encourage the relevant fishery management agencies to become cautious and thus reduce the TACC (Gilman *et al.* 2006, Roche *et al.* 2007). Therefore, the impact that catch depredation by odontocetes has on the management of the affected fishery should be taken into account when determining methods for long term sustainable fishery management.

## 2.6 Bycatch and depredation mitigation strategies

Until recently, few studies have attempted to identify solutions for mitigating operational interactions between odontocetes and longline fishing operations. Most simply flagged promising ideas (*e.g.*, Northridge and Hofman 1999, Visser 2000), while some compiled more detailed accounts of mitigation measures implemented directly by individual longline fishermen and by fisheries in an *ad hoc* and untested manner (*e.g.*, Dahlheim 1988, Secchi *et al.* 2005, Table 2). Independent experimental trials that quantify the effectiveness of potential depredation mitigation strategies in the longline industry remain in their infancy (*e.g.*, Moreno *et al.* 2008, Mooney *et al.* 2009). In contrast, the available literature indicates that the development and implementation of cetacean depredation and bycatch mitigation strategies in other fisheries is more advanced (*e.g.*, purse seine: Gosliner 1999, Hamer *et al.* 2008; gill net: Trippel *et al.* 1999, Barlow and Cameron 2003).

Fishermen, fishery managers, and researchers have proposed a number of techniques for mitigating odontocete depredation and bycatch in longline fishing gear, which can be broadly categorized as (1) behavioural, (2) spatial, (3) acoustic, or (4) physical. Behavioural techniques have been successfully used in active fishing methods, such as trawl (Tilzey *et al.* 2006) and purse seine (Hamer *et al.* 2008), because the fishing gear can be manipulated and monitored during a fishing event to mitigate the likelihood of bycatch. Unfortunately, there is limited scope for the behaviour of fishermen to influence the way longline gear behaves when it is deployed, because it hangs passively in the water column, is not attached to the vessel, and is generally remote and out of sight. Nonetheless, fishermen are able to make decisions about where to fish and for how long to deploy the gear in order to avoid odontocetes, although such practices are yet to be quantified and are unlikely to be implemented voluntarily, especially in the long-term, unless the economic benefits are immediately apparent. The 'move on' tactic has been used by some longline fishermen in a bid to outrun depredating whales, although the success of this strategy seems to be ambiguous at best and is likely to be costly, thus affecting profit margins[1]. A study of this method for avoiding pinniped depredation and bycatch in a trawl fishery found it was only occasionally successful, because depredating individuals were also able to travel long distances to remain with the vessel (Tilzey *et al.* 2006).

Spatial closures, typically known as marine protected areas (MPAs), are designed to spatially separate marine mammal populations and fishing effort so as to reduce the likelihood of bycatch mortalities. However, MPAs that are effective in protecting odontocete populations are difficult to implement, because (1) knowing where to put them is often difficult to determine in the absence of reliable data on odontocete migration and movement patterns (Dulau-Drouot *et al.* 2008), (2) they are often smaller than necessary, due to stakeholder pressure to minimize their impact on fisheries (Klein *et al.* 2008), (3) monitoring compliance by fishermen is difficult due to the lack of capacity and resources (Le Quesne 2009) and (4) quantitative performance

[1] Personal communication: Mark Coker and Niki McCulloch, De Brett Seafood, Queensland, Australia, November 2009 and June 2011, respectively.
[2] Personal communication: Mark Coker, De Brett Seafood, Queensland, Australia, November 2009.
[3] Personal communication: Mark Coker, De Brett Seafood, Queensland, Australia, November 2009; Ueta Faasili,

assessment is hampered by the statistical uncertainties associated with limited and potentially unrepresentative data (Claudet *et al.* 2006). In contrast, the implementation of MPAs to protect pinnipeds from fishing activities has proven easier, because they are central place foragers whose at-sea movements and population trends can be determined with comparative ease (Baylis *et al.* 2008, Shaughnessy *et al.* 2011, Hamer *et al.* 2011).

### *2.6.1 Acoustic technologies*

Acoustic technologies for mitigating odontocete depredation and bycatch in longline fisheries have received the most attention in the literature. In general, high intensity sounds are used to deter depredating odontocetes from approaching fishing gear, while comparatively low intensity sounds are used to alert odontocetes (and other cetaceans) to the presence of fishing gear to prevent them from becoming incidental bycatch.

The majority of the literature in this field reports on ways to mitigate catch depredation and typically focus on four strategies, which are harassment, deterrence, echolocation disruption, and avoidance. Their development has generally been encouraged by fishery stakeholders who hope to reduce the economic impact of depredation on their fishing enterprise. Most effort has focused on the development of acoustic harassment devices (AHDs), which are designed to encourage or force depredating odontocetes to leave the vicinity of the fishing gear (Nowacek *et al.* 2007). In the absence of information on odontocete hearing range and capacity, AHD development seems to have been based on human characteristics. As such, they typically transmit sounds greater than 180 dB (at 1 m from the source), which are beyond the 145 dB level at which human hearing structures are at risk of permanent damage (Price 1981, Nowacek *et al.* 2007).

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Acoustic deterrence devices (ADDs) emit moderately high sounds that are typically lower than 180 dB (at 1 m from the source). Based on similar principals to AHDs, ADDs are designed to annoy depredating odontocetes and encourage them to leave or remain clear of an area, typically where floating structures such as fish pens are located (Dawson *et al.* 1998, Nowacek *et al.* 2007). Unlike AHDs and ADDs, echolocation disruption devices (EDDs) are claimed to prevent depredating odontocetes from accurately determining the location of a caught fish, thus reducing the likelihood of a successful depredation attempt (Mooney *et al.* 2009). Nonetheless, the distinction between ADDs and EDDs remains unclear, because the functional mechanisms that elicit specific behavioural outcomes in depredating odontocetes in response to the sound emitted remains unclear (Jefferson and Curry 1996).

Efforts have also been made to avoid depredation altogether. Passive listening arrays (PLAs) are designed to assist affected fishing vessels in acoustically detecting depredating odontocetes, thus allowing the vessel to leave or move on from a fishing area when individuals are detected in the vicinity (McPherson *et al.* 2008). Unlike AHDs, ADDs, and EDDs, PLAs are not a deterrent mechanism, instead providing fishermen with the ability to reduce the level of physical overlap with depredating odontocetes. Despite the apparent benefits, this technology remains in its infancy. This is partly due to the difficulties associated with confirming the presence or absence of highly mobile odontocetes, with depredation occurring at times when odontocetes are not observed or detected and depredation not occurring at times when odontocetes are observed or detected[2].2 In addition, PLAs have been slow to develop, because the associated structures and equipment are typically complex and expensive (Nielsen and Møhl 2006), they can be damaged by marine predators such as sharks (Johnson *et al.* 1982) and sound interference from the fishing vessel and from the broader marine environment can mask the vocalizations of depredating whales (Thode *et al.* 2007).

*

[2] Personal communication: Mark Coker, De Brett Seafood, Queensland, Australia, November 2009.

An alternative strategy for avoiding depredating odontocetes involves masking or minimizing the sound signature of fishing vessels that are thought to attract depredating odontocetes, so that the individuals involved can no longer detect the vessels presence (McPherson *et al.* 2008). However, identifying a parsimonious suite of sound signature and suppression factors that may assist in mitigating depredation has proven to be logistically and technically challenging.

Acoustic technology has also been used to mitigate incidental bycatch of odontocetes, with the main strategy being to warn individuals of the presence of fishing gear in their vicinity (Kraus *et al.* 1997, Barlow and Cameron 2003). Such devices are referred to as 'pingers'. Their development and implementation has predominantly occurred in association with gill net and drift net fisheries, where incidental bycatch of odontocetes seems to have been greatest (Read *et al.* 2006). A number of studies have shown that pingers significantly reduce incidental bycatch of harbor porpoises (*Phocoena phocoena*) in demersal gill nets (Lien *et al.* 1995, Kraus *et al.* 1997, Trippel *et al.* 1999, Gearin *et al.* 2000) and common dolphins in drift gill nets (Barlow and Cameron 2003). However, other studies have shown that assessing the effectiveness of pingers is difficult, due to the lack of statistical power caused by typically low rates of bycatch, habituation of individuals to the noises made by pingers (which can also lead to the 'dinner bell effect', where individuals are actually attracted by the noise emitted, rather than being warned away or repelled) and a lack of understanding of the processes that lead to bycatch (Dawson *et al.* 1998). Pingers may also be species specific in their application, thus making it difficult to address bycatch effectively, especially in situations where more than one species is involved (Kastelein *et al.* 2006). Despite the potential usefulness of pingers in some fisheries, they are unlikely to be useful in longline fisheries, where most bycatch occurs when depredating odontocetes are actively attempting to remove caught fish from hooks.

*

Despite the considerable efforts to develop acoustic strategies for mitigating depredation and bycatch, to date their successful application has proven difficult. The use of AHDs raises ethical concerns about the effect of high-level noise on odontocetes and the wider marine environment (Johnston and Woodley 1998, Morton and Symonds 2002). One study of the use of ADDs in a gill net fishery suggested that the subsequent reduction in dolphin bycatch may have occurred because the devices deterred the target fish, thus encouraging the dolphins involved to forage elsewhere (Kraus *et al.* 1997). Over time, odontocetes may become habituated to noises emitted by ADDs, EDDs and pingers, thus rendering them ineffective (Jefferson and Curry 1996). These devices may eventually become attractants or a 'dinner bell' to foraging odontocetes, if animals learn to associate the sounds emitted with the presence of palatable fish (Jefferson and Curry 1996, Mooney *et al.* 2009).

Assessing the efficacy of acoustic devices has been hampered by the lack of experimental replication, mainly due to the variety of odontocete species involved, the number of devices currently available in the market place, variations in the configuration of gear used in each fishery and the unique environmental conditions in each coastal or oceanic region (Jefferson and Curry 1996, Dawson *et al.* 1998, Nowacek *et al.* 2007, Kastelein *et al.* 2006). Current technology also constrains the testing and application of acoustic devices, because integral components such as the transponder and batteries are currently large and expensive (McPherson *et al.* 2008, Mooney *et al.* 2009). These problems are further exacerbated by the current lack of understanding of the mechanisms that underpin how noise harasses, deters or warns odontocetes that are close to fishing gear. While there is a need to continue developing and assessing acoustic depredation and bycatch mitigation strategies in longline fisheries, success, and ultimately implementation, will only be possible if a case-by-case approach is adopted, experiments are controlled and replicated, and the necessary components are sufficiently small and cheap for devices to be deployed in large numbers on the gear.

### *2.6.2 Physical technologies*

Physical technologies for mitigating depredation and bycatch of odontocetes have received comparatively little attention. However, recent innovations in developing physical depredation mitigation devices (PDMDs) have proven promising in a Chilean demersal longline fishery for Patagonian toothfish, with the experimental testing of the 'net sleeve' demonstrating that sperm whale depredation could be reduced by 82.8% (Moreno *et al.* 2008, Hamer 2010). Typically, odontocetes are unable to depredate caught fish when demersal gear is deployed on the benthos, with access only possible during the latter stages of the haul. The rigid net sleeve was designed to remain clear of the baited hooks on the benthos during fishing, then descended the branchline under the influence of gravity, thus preventing access to the caught fish by depredating sperm whales (Fig. 4).

Developing solutions for pelagic longline gear is likely to be more challenging than for demersal longline gear. Pelagic longlines are set at much shallower depths (*i.e.*, between 30 m and 300 m from the surface), thus are accessible by most depredating odontocete species throughout the fishing period (Hamer 2010). As such, an effective device would need to be comparatively complex, remaining clear of the baited hook to allow it to function unimpeded. The device must also include a trigger mechanism that is activated by line tension when a target fish becomes caught and attempts to escape. Efforts to solve this problem in other hook-based pelagic fisheries may provide valuable insights. A recent study reported on attempts to mitigate catch depredation by bottlenose dolphins (*Tursiops truncatus*) in the Florida king mackerel (*Scomberomorus cavalla*) troll fishery and found that dolphins were deterred from depredating by a 'metal wire' that moved around in the water next to the caught fish (Zollett and Read 2006). In order to allow the baited hook to fish unimpeded, the metal wire was held clear in a tension sensitive mechanism. When a fish was caught and the tension increased, the metal wire was released and then descended the line towards the caught fish. It was assumed that dolphins were deterred from depredating the caught fish due to fear of physical injury or entanglement.

Although the development of PDMDs (to mitigate both bycatch of and depredation by odontocetes) is in its infancy, they offer a comparatively realistic, applicable and generic approach when compared with the current generation of available acoustic devices. This is because it is possible to manufacture comparatively small and cheap devices that can be placed on each snood immediately adjacent the hook, where depredation and bycatch events take place. Testing their efficacy is also comparatively simple, with the use of rigorous and controlled experimental trials that measure target fish catch rates, and odontocete depredation and bycatch rates. In contrast, the efficacy of acoustic devices is more difficult to determine, because it is not possible to ascertain whether the presence of a device is directly responsible for the results obtained.

Despite the conceptual and experimental advantages associated with developing PDMDs, accommodating the necessary physical complexity of the devices used in pelagic longline fisheries may prove challenging. Unlike acoustic devices that generally have no external moving parts, PDMDs may contain many. A conventional pelagic longline operation involves the deployment of between 1,800 and 3,600 hooks, with a hook deployed every 6–8 s during setting, indicating that the addition of the few extra seconds needed between each hook to attach or remove PDMDs may considerably increase the duration of daily fishing activities (Hamer 2009*b*). In addition, PDMDs need to be small and complex, which may reduce their durability in the harsh marine environment.

*

Pelagic longliners targeting tuna and billfish in the Tropical South Pacific Ocean (TSPO) have unofficially reported depredation by pilot whales and false killer whales for over a decade. In more recent times, a number of anecdotal reports by fishermen have indicated that depredating individuals may be avoiding sections of the longline where the gear has become tangled, mainly as a result of vigorous and prolonged swimming activity by caught billfish and sharks[3]. As such, an approach reminiscent of the Florida king mackerel example may be worthwhile, where structures that simulate or mimic tangles are deployed near a caught fish to both physically and psychologically deter depredating odontocetes. The Australian Government has initiated a study of this nature, which is currently in the developmental phase, although extensive sea trials on commercial longline vessels will soon commence in the TSPO and in the Indian Ocean (Hamer 2011, Fig. 5).

The success of PDMDs is dependent on the level of industry implementation, which is more likely to be associated with voluntary uptake, rather than mandatory or enforcement means. Mandating or enforcing the use of PDMDs that have no obvious benefit to the fishery would likely result in the need for costly monitoring activities and in some level of noncompliance, both of which are unfavourable outcomes. Specifically, fishermen are unlikely to purchase and implement this technology if the cost of doing so is more than the increases in revenue associated with reduced catch damage, suggesting the aim should be to ensure a cost-benefit analysis will work in the favour of the fisherman. This is especially important when considering the large number of illegal, unregulated, and unreported (IUU) pelagic longline fisheries around the world that avoid conventional management regulations (Food and Agriculture Organization 2001, Baker *et al.* 2006, Lukoschek *et al.* 2009). As such, PDMD development should not only focus on efficacy and durability, but also on cost minimisation.

[3] Personal communication: Mark Coker, De Brett Seafood, Queensland, Australia, November 2009; Ueta Faasili, Samoa Ministry of Fisheries, Apia, Samoa, February 2010; Tom Mayo, Solander Fisheries, Suva, Fiji, August 2011.

## 2.7 Summary and future directions

The literature summarized here indicates that odontocete bycatch and depredation in longline fisheries is widespread, involving many fisheries and many odontocete species and populations. Mitigating this problem is becoming a higher priority for all stakeholders, especially as longline fishery profit margins dwindle and as competition and conflict between odontocetes and longline fisheries increase. Nonetheless, the problem of depredation may be overstated by some fishermen who attribute poor catch performance to odontocete depredation, when the real cause may be poor operational decisions and thus poor catch performance, or incorrect assignment of depredation by other taxa (*i.e.*, sharks, fish or squid). In contrast, the problem may be understated for odontocetes, because small and as yet unidentified populations that include depredating individuals may be at risk of decline with the loss of only a few individuals (Leatherwood *et al.* 1983, Beissinger and McCullough 2002, Miller 2007). An earlier review of this issue indicated that the successful mitigation of bycatch and depredation can only occur with changes in longline fishing practices, although it lamented that little had been done thus far to identify and test potential solutions (Gilman *et al.* 2006). Given the volume of information summarized here, the geographic extent of operational interactions and the efforts made to develop mitigation strategies to date, stakeholders are strongly encouraged to prioritize this problem without delay, with a view to conserving affected odontocete populations and sustaining longline fisheries.

Both acoustic and physical mitigation technologies have shown some promise, although both appear have inherent problems that may hinder their development and implementation. Acoustic mitigation tools are simple in their application, although cumbersome and inadequate in their function and complex in their assessment. For example, the size of the batteries and transponders currently available are probably too big for most applications, while the types and levels of noises that deter specific odontocete species remain unclear and testing the efficacy of those noises in a highly dynamic marine environment is difficult. By comparison, PDMDs are simple in their function and assessment, although their practical application and implementation may be challenging if maintenance and per-unit cost are high. Perhaps the most important aspect of ensuring the success of any bycatch and depredation mitigation strategy, whether it be acoustic, physical, or any other form, is the need to keep purchase and implementation costs below the economic gains associated with increased catch revenue. Although some fishery management agencies may opt for mandatory implementation of these technologies, the reality is that noncompliance is likely to be widespread if there are no economic benefits for the affected fishery. Longline fishermen involved in IUU activities are likely to monitor developments in regulated or managed fisheries and will only adopt mitigation technologies if there is a perceived economic benefit. Therefore, stakeholders involved in the development of PDMDs and other depredation and bycatch mitigation technologies should aim to minimize costs in order to increase the likelihood of voluntary implementation by the affected fishery regardless of its management status.

Individual longline operations and fisheries around the globe are likely to face a diversity of situations, such as differences in (1) the odontocete species they interact with and bycatch rates, (2) target fish depredation rates, catch rates and value, and (3) overall operational costs (including repayments, fuel, wages, bait, *etc*.). Given that a single panacea to this diverse problem is unlikely to emerge, fishermen and fishery managers are encouraged to maximize the chance of mitigating odontocete bycatch and depredation by using a suite or 'toolbox' of mitigation strategies, such as (1) acoustic, (2) physical, (3) fishermen behaviour, and (4) MPAs (*e.g.*, Dahlheim 1988, Gilman *et al.* 2002, Gilman *et al.* 2006, Campbell and Cornwall 2008).

*

The development and implementation of acoustic and physical mitigation methods has attracted the interest of a broad stakeholder base. At the policy end of the spectrum, regional partnerships and agreements are deemed necessary for facilitating necessary research and for securing necessary funds. For example, the joint tuna regional fishery management organizations (T-RFMOs) have implemented the 'Kobe bycatch process' to deal with issues of whale bycatch and depredation. However, a number of delegates at the *Kobe II Bycatch Workshop* (Brisbane, Australia) in 2010 criticized the process, citing (1) a lack of consensus about fundamental terminology (*e.g.*, what constitutes bycatch?), (2) the overcomplicated and slow process, and (3) the misguided focus on documentation rather than problem solving (Kobe II 2010). At the operational end of the spectrum, researchers appear to have made extensive inroads toward finding solutions, by focusing a growing body of knowledge on the subject of mitigating odontocete bycatch and depredation (*e.g.*, Gilman *et al.* 2006, Zollett and Read 2006, Moreno *et al.* 2008, McPherson *et al.* 2008, Hamer 2011). Nonetheless, researchers have acknowledged that the conceptualization and development of some solutions may lay with the fishermen whose knowledge, experience and enthusiasm should not be underestimated or undervalued (Gilman *et al.* 2006). Regardless of where one might sit on the policy and operations spectrum, it is clear that greater resources and commitment need to be focused on assisting fishermen and researchers to address this issue at the vessel, preferably at the hook. If this does not occur soon, it should be expected that operational interactions (depredation and bycatch) between odontocetes and longline fisheries will continue or increase, thus casting uncertainty on the future of affected longline fisheries and on the fate of affected odontocete populations.

Table 1 Summary of operational interactions (*i.e.*, catch depredation and bycatch) between odontocetes and pelagic and demersal longline fishing gear, inferred from or quantified in the literature reviewed.

| Whale species involved | Fishery details: species targeted | Fishery details: region of interaction | Fishery details: gear configuration | Catch depredation details: % of sets affected [3] | Catch depredation details: % of catch damaged [3] | Whale bycatch details: # of whales Hooked | Whale bycatch details: rate (animals/set) | Source author(s) | year |
|---|---|---|---|---|---|---|---|---|---|
| ? | ? | IO | ? | | <55 | | | Sivasubramaniam | 1964 |
| KW, FKW | T | IO | P | | | | | Mitchell | 1975 |
| GTB | ? | TC | ? | | | 2 [6] | | Watson | 1981 |
| CD | S | FAC | P | | | 2 [6] | | Duguy & Hussenot | 1982 |
| SW | | CM | P | | | 1 | | Di Natale & Mangano | 1983 |
| KW | SF | PWS | D | | 25 | | | Matkin | 1986 |
| KW | SF | BS, PWS | D | (15-25) | | | | Dahlheim | 1988 |
| KW | SF, GT, AF | BS, GA | D | | (13-45) | | | Yano and Dahlheim | 1995 |
| SW, KW | PT | SG | D | 93 | >90 | 2 [6] | 0.07 | Ashford *et al.* | 1996 |
| KW | T, SwF | SB | P | | (50-100) | | | Secchi & Vaske | 1998 |
| KW | SS, BET | NZ | D | | 5-10 | | | Visser | 2000 |
| SRW | | | P | | | 3 [7] | | Best *et al.* | 2001 |
| SW | SF | GA | D | | 23 [5] | | | Straley *et al.* | 2002 |
| KW, B | T, SwF | EA | P | | | 2 [6] | | Shaughnessy *et al.* | 2003 |
| SW, KW | PT | SC | D | 16 | 3 (0-100) | | | Hucke-Gaete *et al.* | 2004 |
| SW, KW | PT | SG | D | 13 | | | | Perves *et al.* | 2004 |
| KW, PW | BET, L | SAus | D | 6-80 [4] | | | | AFMA [13] | 2005 |
| KW, FKW, B, D | T, SwF | EA | P | | | 5 [8] | | Bell *et al.* | 2006 |
| SW, KW | PT | SG, PEI | D | | >50 | | | Kock *et al.* | 2006 |
| BD | KM | F | P | | 6-20 | | | Zollett & Read [14] | 2006 |
| KW | T, SwF | SB | P | | 12 (1-47) | | | Dalla Rosa & Secchi | 2007 |
| KW | T, SwF, S | SA | | | 0.50 | | | Williams *et al.* | 2007 |
| KW | PT | CA | D | | 42 | | | Roche *et al.* | 2007 |
| Various [1] | | | | | | 67 [9] | 0.003 [11] | Forney & Kobayashi | 2007 |
| FKW | T, SwF, S | B, AA | P | (1-9) | <9 | 2 [6] | | Hernandez-Milian *et al.* | 2008 |
| FKW | SwF | A, IO, P | | 2 | 4-16 | 18 [10] | 0.002 [11] | Ramos-Cartelle & Mujeto | 2008 |
| SW | SF | BS, GA, AI | D | | <1 | | | Sigler *et al.* | 2008 |
| SW, KW | PT | SC | D | | 0.36 | | | Moreno *et al.* | 2008 |
| SW, KW | PT | CA | | | 41 | | | Tixier *et al.* | 2009 |
| CD, BD, SD | Various [2] | IC | D, P | | 40 | | | Lauriano *et al.* | 2009 |
| FKW, PW | T, BF | CS | P | <16 | <10 | 3 [6] | 0.231 | Hamer | 2009*b* |
| DD | D ,S | PC | P | | | 1 | 0.05 [12] | Mangel *et al.* | 2010 |

**Whale species abbreviations**

| | |
|---|---|
| B | Unidentified baleen whale species (Mysticeti) |
| BBW | Blainsville's beaked whale (*Mesoplodon densirostris*) |
| BD | Bottlenose dolphin (*Tursiops truncatus*) |
| BW | Bryde's whale (*Balaenoptera edeni*) |
| CD | Common dolphin (*Delphinus delphis*) |
| D | Unidentified small toothed whale species (Odontoceti) |
| DD | Dusky dolphin (*Lagenorhynchus obscurus*) |
| FKW | False killer whale (*Pseudorca crassidens*) |
| GTB | Ginko-toothed beaked whale (*Mesoplodon ginkgodens*) |
| HW | Humpback whale (*Megaptera novaengliae*) |
| KW | Killer whale (*Orcinus orca*) |
| PW | Pilot whale (*Globicephala* spp.) |
| RD | Risso's dolphin (*Grampus griseus*) |
| SD | Striped dolphin (*Stenella coeruleoalba*) |
| SPD | Pantropical spotted dolphin (*Stenella attenuata*) |
| SRW | Southern Right Whale (*Eubalaena australis*) |
| SW | Sperm whale (*Physeter macrocephalus*) |

**Further explanation**

1 PW, FKW, SPD, BD, BBW, RD, SW, BD, CD and HW.
2 Unspecified fish species.
3 Values are averages or estimates; values in parentheses are ranges.
4 6% of sets affected, calculated from industry data; 80% of sets affected, derived from anecdotal information from fishers.
5 Inferred from a reduction in the catch rate of the targeted fish.
6 Dead animals recorded.
7 Entanglement mortalities.
8 5 animals hooked; 2 dead (1 KW and 1 D) and 3 released alive.
9 67 hooked; 7 dead (2 PW, 2 FKW, 1 SpD, 1 BD and BBW) and 60 released alive.
10 18 animals hooked; proportion dead and released alive not specified.
11 Derived retrospectively from figures presented in the results of the study.
12 In addition, harpooning of dolphins for bait was occasionally observed.
13 Australian Fisheries Management Authority.
14 Study of a troll fishery – included here due to the relevance of the depredation mitigation strategy to longline fishing.

**Fish species abbreviations**

| | |
|---|---|
| AF | Arrowtooth flounder (*Atheresthes stomias*) |
| BET | Blue-eye trevalla/bluenose (*Hyperoglyphe antarctica*) |
| BF | Billfish (Istiophoridae & Xiphiidae) |
| D | Dorado (*Coryphaena hippurus*) |
| GT | Greenland terbot/halibut (*Reinhardtius hippoglossoides*) |
| KM | King mackerel (*Scomberomorus cavalla*) |
| L | Unspecified ling species (*Genypterus* spp.) |
| PT | Patagonian toothfish (*Dissostichus eleginoides*) |
| S | Unspecified shark species (Selachimorpha) |
| SF | Sablefish (*Anoplopoma fimbria*) |
| SS | School shark (*Galeorhinus galeus*) |
| SWF | Swordfish (*Xiphias gladius*) |
| T | Tuna (*Thunnus* spp.) |

**Region abbreviations**

| | |
|---|---|
| A | Atlantic |
| AA | Azores Archipelago |
| AI | Aleutian Islands |
| B | Brazil |
| BS | Bering Sea |
| CA | Crozet Archipelago |
| CM | central Mediterranean |
| CS | Coral Sea |
| EA | eastern Australia |
| F | Florida |
| FAC | French Atlantic coast |
| GA | Gulf of Alaska |
| IC | Italian coast |
| IO | Indian Ocean |
| NZ | New Zealand |
| P | Pacific |
| PC | Peruvian coast |
| PEI | Prince Edward Islands |
| PWS | Prince William Sound |
| SA | South Africa |
| SAus | southern Australia |
| SB | southern Brazil |
| SC | southern Chile |
| SG | South Georgia |
| TC | Taiwanese coast |

Table 2 Summary of methods previously considered or trialed by fishers and researchers to mitigated catch depredation by whales from longlines.

| Method: category and type | Method: Description | Result: success/failure | Problems: realized or perceived | Source |
|---|---|---|---|---|
| **Physical** | | | | |
| Net sleeve | Branch line mounted. Prevents access. Passively drops over hooks and caught fish during hauling. | Success + | • Intelligent animals have learned to damage tail of fish<br>• Refinements needed – longer sleeve | 1 |
| Metal wire | Line mounted. Flaps about to deter cetacean. Descends troll line when fish is caught. | Success * + | • Dependent on whales being deterred by the presence of streamers. | 2 |
| Streamers/tangles | Snood mounted. Flaps about to deter cetacean. Descends snood when fish caught. | Pending outcome + | • Dependent on whales being deterred by the presence of streamers.<br>• Requires complex device, so may have maintenance problems. | 3,7,8 |
| **Chemical** | | | | |
| Lithium chloride / ether | Elicits vomit response. Mounted near hook. Activated when fish caught. | Not trialed | • Unknown health issues for depredating whales and humans.<br>• Potential ethical issues. | 3,4,8 |
| Stress / decay marker | Elicits escape/exit response. Mounted near hook. Activated when fish caught. | Not trialed | • May dissipate too quickly, or have adverse effects over wide area. | 4,8 |
| **Electrical** | | | | |
| Stinger | Snood mounted. Deployed when fish caught and activated when cetacean approaches. | Pending outcome + | • Potential ethical issues for cetaceans and safety issues for crew.<br>• May be difficult to maintain. | 3,4 |
| **Visual** | | | | |
| Bubble screen | Interferes with visual sense. | Not trialed | • Logistically difficult to achieve over wide area. | |
| **Acoustic** | | | | |
| Detection | Use of listening devices to pick up echolocation signals from cetaceans in the area. | Limited success + | • Results are often ambiguous and inconclusive.<br>• Works over an insufficient distance. | 4,7 |
| Predator playback | Use of predator noises to elicit escape response such as killer whale calls to deter pilot whales. | Not trialed | • Individuals may become habituated, making them vulnerable.<br>• Works over insufficient distance. | 4 |
| Masking / disruption | Producing predominant 'white noise' to mask noises produced by vessel activities. | Initial success | • Trialed on a captive animal only.<br>• Demonstrated learning by individual reduced device performance. | 4,6 |
| Harassment | Annoying and potentially damaging sound forces cetaceans to leave the area. | Unsuccessful + | • May cause hearing damage and stranding.<br>• May have adverse effects on other animals. | 4 |
| Accessory skiffs | Acoustic novelty draws cetaceans away from fishing gear. | Not trialed | • Would only work on demersal longlines where line comes up to boat.<br>• Logistically difficult to achieve for pelagic longlines. | 4 |
| Quiet operations | Modify vessels to make less noise. | Initial success | • Individuals may learn to detect signatures in background noise. | 3,5,8 |
| Explosives / seal bombs | Loud noise causes flight response. | Unsuccessful | • May cause hearing damage and stranding.<br>• May have adverse effects on other animals. | 4 |
| **Behavioral** | | | | |
| Operant conditioning | Behavioral modification using signal cues. | Not trialed | • Requires high proportion of animals in the population to learn. | 4 |
| Blank sets | Gear set without baits to confuse whales. | Unsuccessful | • Depredating individuals soon learned to search for baited sets. | |
| **Management** | | | | |
| Spatial closures | Away from areas frequented by depredating cetaceans. | Not trialed | • Moves effort to a different location – may cause other problems.<br>• Often puts effort outside prime fishing ground. | 7,8 |
| Temporal closures | Away from areas frequented by depredating cetaceans at certain times of the year. | Not trialed | • Moves effort t a different time of year – may cause other problems.<br>• Often puts effort outside prime fishing period. | 7,8 |
| Move fishery | Away from traditional fishing grounds to areas not frequented by depredating cetaceans. | Limited success | • Large volume of fuel to move >60 nm.<br>• Often puts vessels outside prime fishing ground. | 4,9 |
| Change target species | To a species thought to be unattractive to depredating cetaceans. | Mixed results | • Alternative species often more difficult to catch or less valuable.<br>• Depredating whales learn to take advantage of new food source. | 4,8 |
| Change time of fishing | Fish at night instead of during the day. | Unsuccessful | • May only be effective for species that only feed during the day. | 4 |
| Change depth of set | Out of depth range of depredating cetaceans. | Limited success | • May also put gear beyond depth of target fish species. | 8,9 |
| Change gear type | Use pots to catch the fish instead of longlines. | Limited success | • Possible only in demersal fisheries<br>• Often results in reduced catch. | 4,7 |
| Culling | Shooting or harvesting of cetaceans. | Not trialed | • Illegal and unethical. | 4 |

**Information source**
1 Moreno *et al.* 2008
2 Zollett and Read 2006*
3 Hamer 2009*b*
4 Dahlheim 1988
5 AFMA 2005

**Information source (continued)**
6 Mooney *et al.* 2009
7 McPherson *et al.* 2008
8 Gilman *et al.* 2006
9 Tixier *et al.* 2009

**Further explanation**
* Study of a troll fishery – included here due to the relevance of the depredation mitigation strategy to longline fishing.
\+ Outcome based on experimental trials.

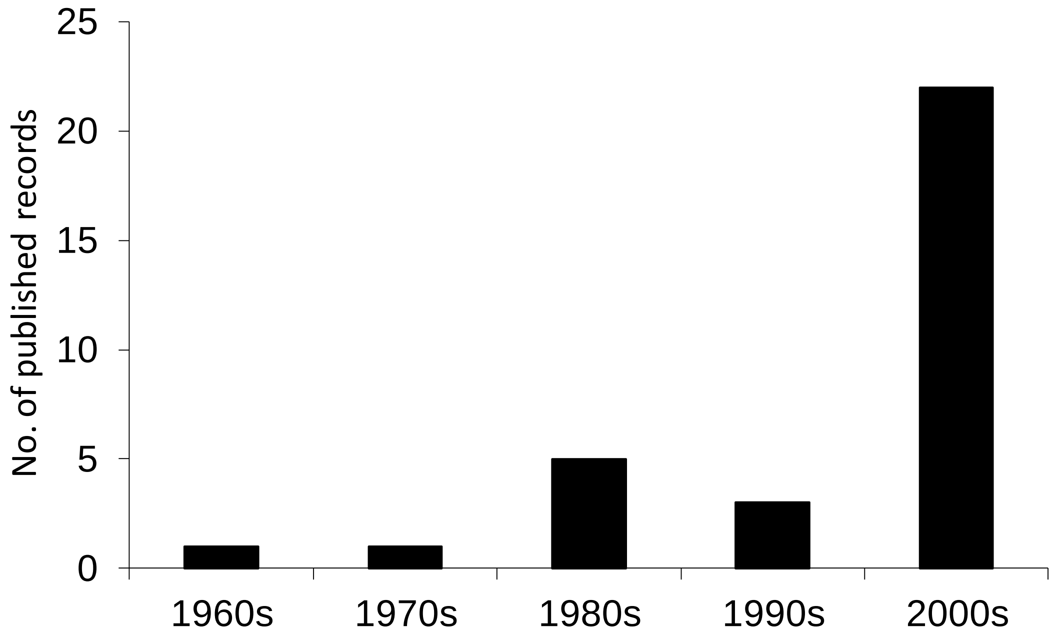


Figure 1 Decadal summary of the number of reviewed reports of operational interactions between odontocetes and longline gear over the last 50 years (the 2000s includes one study published in early 2010).

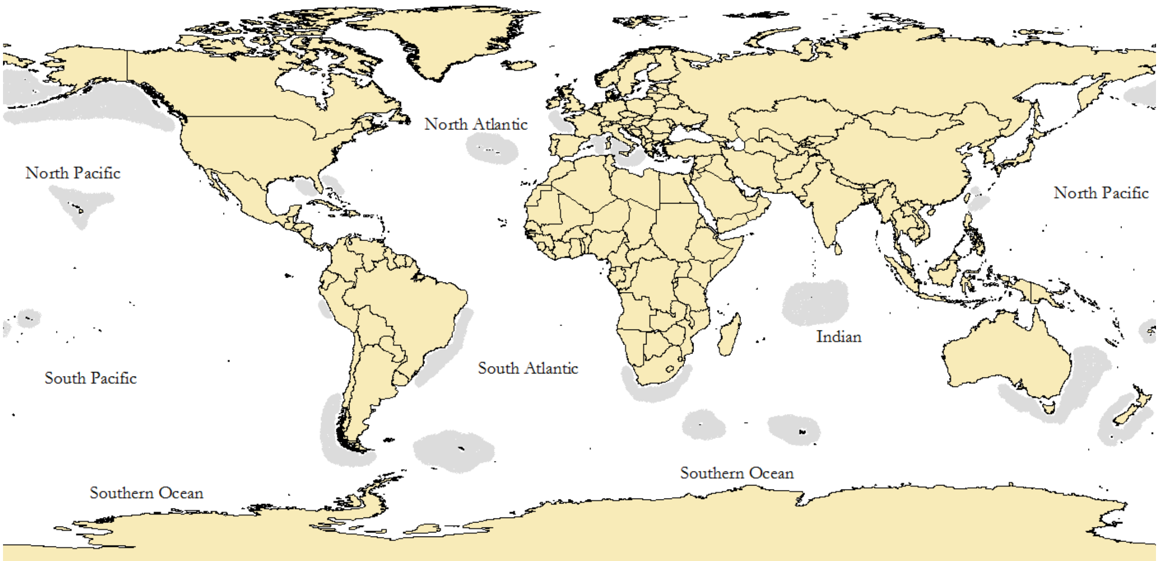


Figure 2 Geographic distribution of operational interactions (inferred and quantified) between odontocetes and longline gear (grey areas) in accessible reports.

NOTE:
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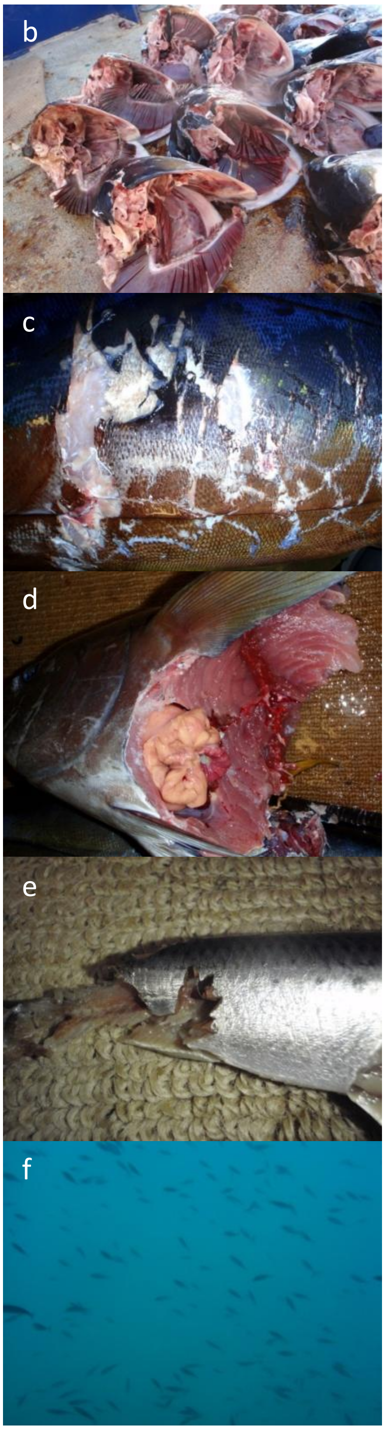


Figure 3 (a) False killer whale caught on a pelagic longline hook in Hawaiian waters (Source: National Marine Fisheries Service), (b) albacore depredated by odontocetes, with the torso completely removed from behind the gill plates, (c) odontocete tooth lacerations on torso of depredated albacore, (d) for comparison, damage caused by depredating shark, showing much cleaner removal of torso, (e) small bight marks on a sardine (*Sardinops sagax*) used for bait, probably caused by small depredating fish, (f) large numbers of small fish in the vicinity of longline fishing gear that may be involved in bait depredation.

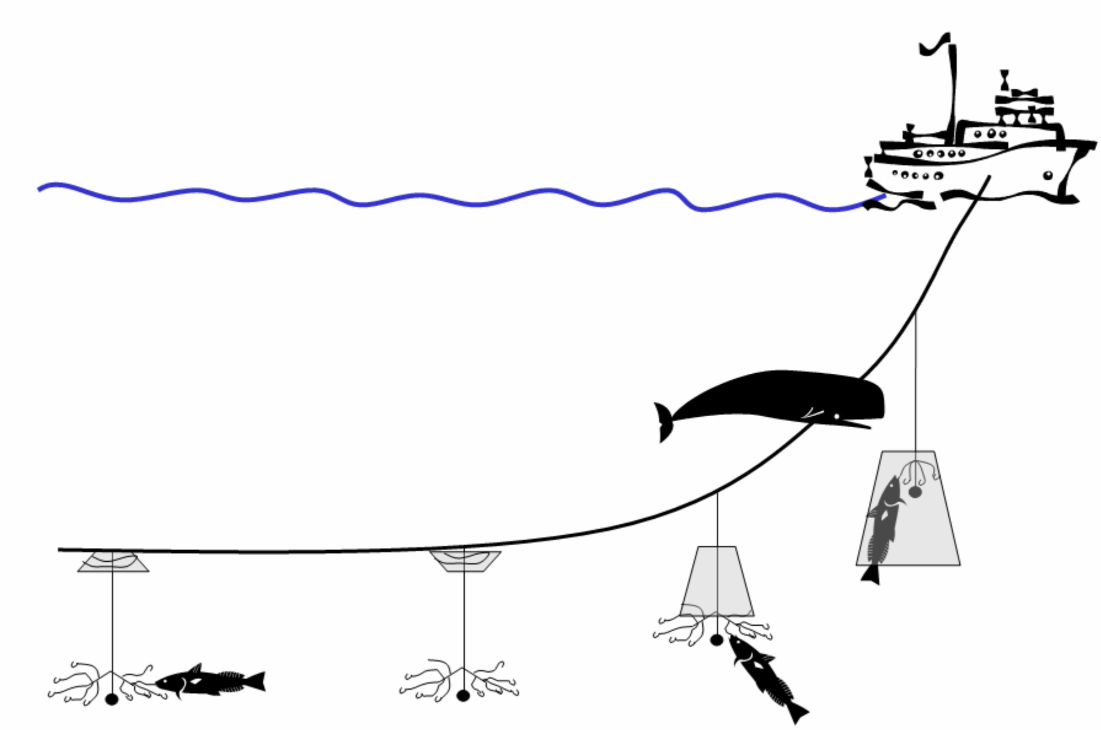

Figure 4 Schematic 'time-lapse' view of net sleeve operation, showing the protection of the fish and physical deterrence of a sperm whale during the haul (With permission: Carlos A. Moreno).

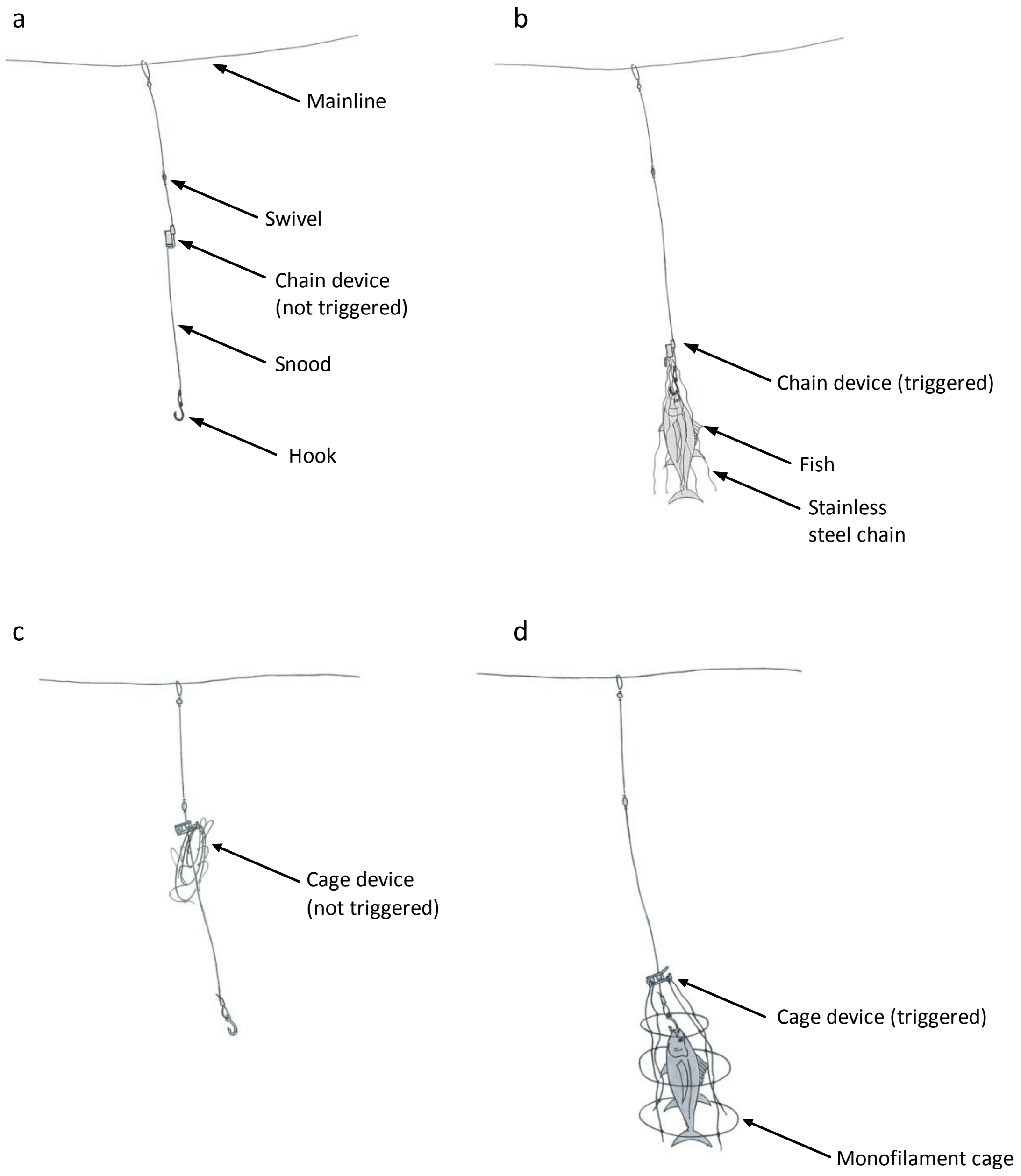


Figure 5 Schematic diagram of the *Chain device* and of the *Cage device* (a,c: not triggered; b,d: triggered) currently under development by the Australian Government and soon to be trialed in the Pacific and Indian Oceans. Before the devices are triggered by the tension of a caught fish, they remain clear of the baited hook and close to the mainline or swivel. Upon being triggered, the devices release the streamers or cage and then descend the snood toward the caught fish, eventually enveloping it.

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# Physical and psychological deterrence strategies for mitigating odontocete by-catch and depredation in pelagic longline fisheries

*Experience is a cruel teacher: it gives you the exam, with the lesson following often some considerable time after.*

Sir Winston Churchill

## Statement of Authorship

| | |
|---|---|
| Title of publication: | Physical and psychological deterrence strategies for mitigating odontocete by-catch and depredation in pelagic longline fisheries |
| Publication status: | Published, May 2012 |
| Publication details: | Progress report to Pacific Island Forum Fisheries Agency (FFA), Worldwide Fund for Nature (WWF) South Pacific, and Pacific Islands Tuna Industry Association (PITIA). Australian Marine Mammal Centre (AMMC), Australian Department of Sustainability, Environment, Water, Population and Communities (DSEWPaC). 47pp. |

### Author Contributions

By signing the Statement of Authorship, each author certifies that their stated contribution to the publication is accurate and that permission is granted for the publication to be included in the candidate's thesis.

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Hamer, D.J. & Childerhouse, S.J. (2012) Physical and psychological deterrence strategies for mitigating odontocete by-catch and depredation in pelagic longline fisheries.
*Progress report to Pacific Islands Forum Fisheries Agency (FFA), World Wildlife Fund (WWF) South Pacific and Pacific Islands Tuna Industry Association ( PITIA).*

**Measurement, management and mitigation of operational interactions between the South Australian Sardine Fishery and short-beaked common dolphins (*Delphinus delphis*)**

NID OES

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Second Penrhyn lockout, 1900-1903

## Statement of Authorship

Title of publication: Measurement, management and mitigation of operational interactions between the South Australian Sardine Fishery and short-beaked common dolphins (*Delphinus delphis*)

Publication status: Published, October 2008

Publication details: *Biological Conservation* 141, 2865-2878

### Author Contributions

By signing the Statement of Authorship, each author certifies that their stated contribution to the publication is accurate and that permission is granted for the publication to be included in the candidate's thesis.

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| Signature: | Date: 27 October 2012 |

## 4.1 ABSTRACT

This study arose from recommendations given in response to a legislated ecological assessment of the South Australian Sardine Fishery in 2004, urging it to: (i) attempt to mitigate operational interactions with marine mammals if excessive levels were detected; and (ii) improve the accuracy of their reporting of these events. An initial observer program revealed high rates of encirclement and mortality (1.78 and 0.39 dolphins per net-set, respectively) of short-beaked common dolphins (*Delphinus delphis*). This equated to an estimate of 1728 encirclements and 377 mortalities across the entire fleet over the same period. The average time taken for fishers to respond to encirclements was 135.93 ± 3.72 min and 21.3% of encircled animals subsequently died. During that time, fishers only reported 3.6% of encirclements and 1.9% of mortalities recorded by observers.

A code of practice (CoP) was subsequently introduced aimed at mitigating operational interactions. A second observer program revealed a significant reductions in the observed rates of dolphin encirclement (0.22; down 87.3%) and mortality (0.01; down 97.1%) with an estimate of 169 and eight, respectively. The average time taken for fishers to respond to dolphin encirclements also reduced to 16.33 ± 4.67 min (down 76.9%) and the proportion of encircled animals that subsequently died reduced to 5.0%. Agreement between industry reports and observer records improved, with the fishery reporting 57.9% and 58.9% of the rate of encirclements and mortalities, respectively, recorded by observers.

A number of avoidance and release strategies in the CoP may have been responsible for these improvements. In particular, fishers were required to delay or relocate their activities if dolphins were observed prior to fishing and to release encircled dolphins immediately or abort the fishing event if release procedures were unsuccessful. Future improvements to the CoP include: (i) improved response times when an encircled dolphin is detected; (ii) better use of behavioural

cues for deciding when to abort a net-set; (iii) ceasing fishing during rough weather; and (iv) continuing to increase reporting accuracy by fishers. It is also recommended that the abundance, movements and boundaries of the common dolphin population in the region be determined, so that the impact of fishing activities on their status can be established.

## 4.2 ACKNOWLEDGEMENTS

We sincerely thank the following organisations and individuals for their assistance with this study: all licence holders, skippers and crews of the SASF who provided access to vessels and volunteered information (in particular: Branko Sarunic, Stan and Dinko Lukin, Tony Santic, Peter White, Graham Tapley and Jo Puglesi); Keith Rowling, Alex Ivey, Matt Hoare and Shane Roberts (SARDI Aquatic Sciences) for collecting observer data during the second observer program; Malcolm Knight, Angelo Tsolos and Emily Thompson (SARDI Statistics) for collating and providing access to SASF logbook data; Steve Shanks and Craig Noell (PIRSA Fisheries, former and current SASF Managers, respectively) and Christian Pyke (SASF, current South Australian Pilchard Fisherman's Association president) for providing assistance and advice in relation to implementing the CoP; Justine Kenyon-Benson (PIRSA Compliance) for assistance with determining vessel movements and fishing effort; Stephen Mayfield, Tony Fowler, Qifeng Ye, Paul Rogers (SARDI Aquatic Sciences) and Peter Shaughnessy (South Australian Museum), plus three anonymous reviewers for assisting in improving drafts. This study was jointly funded by the SASF, PIRSA and SARDI.

## 4.3 INTRODUCTION

### *4.3.1 Dolphin interactions with purse-seine fisheries*

There is now sufficient evidence confirming the occurrence of marine mammal by-catch in numerous trawl, gill-net, longline and purse-seine fisheries in many parts of the world (Northridge, 1984, 1991; Francis and Orbach, 1992; Silva and Best, 1996; Gosliner, 1999; Hale et al., 1999; Trippel et al., 1999; Staunton-Smith and Ward, 2000; De Master et al., 2001; Kemper and Gibbs, 2001; Barlow and Cameron, 2003; Shaughnessy et al., 2003; Bell et al., 2006; Hamer and Goldsworthy, 2006; Read et al., 2006). However, few have described the nature of these encounters, or quantified them in any detail. Operational interactions occur when both marine mammals and commercial fishing activities converge on the same spatially retracted school of fish (Hamer and Goldsworthy, 2006). In doing so, marine mammals may come into direct physical contact with the fishing gear, which may ultimately result in their injury or death (Beverton, 1985; Shaughnessy et al., 2003).

Operational interactions between dolphins and purse-seine fisheries have received considerable attention in the available literature (Francis and Orbach, 1992; Di Natale and Notarbartolo di Sciara, 1994; Gosliner, 1999; Hale et al., 1999; Staunton-Smith and Ward, 2000). The most notable example is the millions of spotted (*Stenella attenuata*), spinner (*S. longirostris*) and common (*Delphinus* spp.) dolphins incidentally killed by the tuna purse-seine fishery in the eastern tropical Pacific (ETP) between the 1960s and 1990s, with the annual kill peaking at 457,903 in 1969 (Francis and Orbach, 1992; Joseph, 1995; Wade, 1995; Gosliner, 1999; Archer et al., 2001, 2004). Dolphins indicate the presence of tuna in the eastern tropical Pacific because the two are closely associated, thus resulting in the intentional targeting of dolphins during 41.7% of the 18,609 net-sets conducted by the 132 United States registered vessels in 1974 (Joseph, 1995; Gosliner, 1999). The US Marine Mammal Protection Act was introduced in 1972, partly in response to this issue. An observer program was implemented during the early 1980s and the practice of deliberately setting purse-seine nets around dolphin pods was subsequently prohibited (Gosliner, 1999). A 'back-down' procedure was also introduced to facilitate the escape of encircled dolphins, by creating an escape route between the top of the submerged net and the surface of the water. By 1983, dolphin mortalities had declined to 8513, or 98.1% when compared with the 1969 peak (Gosliner, 1999).

In Australia, the only published reports of operational interactions between dolphins and purse-seiners originate from a developmental pilchard fishery in southern Queensland, during the mid 1990s (Hale et al., 1999; Staunton-Smith and Ward, 2000). An independent observer program recorded 77 encirclements and 9 mortalities from 63 net-sets, producing an encirclement rate of 1.22 per net-set and a mortality rate of 0.14 per net-set (Hale et al., 1999; Staunton-Smith and Ward, 2000). Encirclements were defined as animals swimming freely within the pursed net and mortalities were defined as those animals that ultimately died. A working group comprising industry representatives and researchers was established to address the issue. They recommended changes to fishing practices be introduced, including improvements to avoidance and release procedures. In particular, it was suggested that encircled dolphins should be released by lowering a portion of the corkline to create an opening, or by aborting the fishing operation entirely before dolphins became stressed and died (Staunton-Smith and Ward, 2000). However, a blanket prohibition on purse-seine fishing in Queensland waters was declared before the effectiveness of these measures could be tested (State of Queensland, 2000; Staunton-Smith and Ward, 2000).

### *4.3.2 South Australian Sardine Fishery*

The South Australian Sardine Fishery (SASF) was established in 1991 to provide food for wild-caught southern bluefin tuna (*Thunnus maccoyii*), ranched in sea cages off Port Lincoln, South

Australia (Fig. 1). Most of the sardine (*Sardinops sagax*) catch is taken from southern Spencer Gulf, although some fishing occurs west of Coffin Bay and off the north coast of Kangaroo Island. Catches in the fishery increased from 3241 t (number of net-sets unknown) in 1994 to 39,839 t (1275 net-sets) in 2005, making it the largest fishery by weight in Australia. There are no spatial or temporal closures and the total allowable commercial catch (TACC) is currently set for each calendar year (Rogers and Ward, 2006).

The sardine fishery is a typical, modern purse-seine fishery. Most fishing occurs at night or at twilight. About 14 vessels operate under licence and although they vary between 18 and 42m in length, they all use nets that are 500–700m in length and are between 40 and 70m deep, with mesh size ranging from 14 to 22 mm. The floatline holds the top of the net at the surface, while the leadline causes the bottom of the net to sink rapidly, thus creating a 'curtain'. Once a target school is selected, it is encircled with the curtain of net and the leadline is pursed to prevent the escape of the catch (Fig. 2). The bulk of the net is then hauled aboard, until the catch is brought alongside the vessel and pumped into onboard holding tanks.

### *4.3.3 Statutory protection of marine mammals in South Australia*

Marine mammals in South Australian waters are protected under both South Australian state and Australian Commonwealth legislation (Bache, 2003). The relevant state legislation includes the National Parks and Wildlife Act 1972, the Fisheries Act 1982 and the Wilderness Protection Act 1992, which specifically prohibit the intentional or negligent killing and exploitation of marine mammals. The Commonwealth Environment Protection Biodiversity Conservation Act 1999 (EPBC Act), which is administered by the Commonwealth Department of the Environment, Water, Heritage and the Arts (DEWHA), also prohibits the intentional killing or exploitation of any listed marine species, including dolphins, in both South Australian and Australian

Commonwealth waters. All major Australian fisheries must now undergo an environmental assessment under the guidelines for the ecologically sustainable management of fisheries, pursuant to the EPBC Act, and address any subsequent recommendations by DEWHA before the required exemption to remove or export a native species is granted.

An environmental assessment of the sardine fishery was undertaken by the Department of Primary Industry and Resources South Australia (PIRSA; the manager of the fishery) in September 2004, pursuant to the EPBC Act, to identify possible effects of its activities on the wider marine ecosystem (Shanks, 2004). The fishery was subsequently given approval by DEWHA, although PIRSA were specifically required to address a number of recommendations for improving the management arrangements of the fishery (Tailby, 2004). Two of these recommendations stated that the fishery should: (i) establish a mechanism that ensures operational interactions with marine mammals are reported accurately; and (ii) develop appropriate mitigation measures if a significant level of operational interactions are occurring.

#### *4.3.4 Development of a code of practice for dolphin by-catch mitigation*

A study to address these recommendations was implemented by the South Australian Research and Development Institute (SARDI) in November 2004. An observer program was initiated to assess the accuracy of logbook data and measure interaction rates. Rates of encirclement and mortality of short-beaked common dolphins (Delphinus delphis) recorded by observers were found to be much higher than those reported in logbooks. The fishery was then closed as a precautionary measure during August and September 2005, to prevent further interactions with dolphins, while a code of practice (CoP) was finalised (South Australian Pilchard Fisherman's Association, 2005).

*

A working group was established that included industry representatives, licence holders, fishers, researchers and fishery managers, with a mandate to improve reporting accuracy and mitigate future operational interactions with dolphins through the CoP. The underpinning principles were that it must: (i) significantly reduce operational interactions with dolphins; (ii) facilitate improvements in fishing practice through ongoing development based on input from all stakeholders; (iii) be sufficiently flexible to be safely and practically applied on all vessels under all conditions; and (iv) be cost-effective to implement. The CoP aimed to mitigate operational interactions between dolphins and the fishery through:

- *Early detection*. At least one crewmember was required to determine the presence/absence of dolphins before and during each fishing event and to immediately report any sightings to the skipper.
- *Avoidance*. The skipper was required to delay or relocate the fishing event if dolphins were detected before commencing fishing.
- *Swift action*. The skipper was required to initiate release procedures without delay when encircled dolphin(s) were detected during the fishing event.
- *Abort fishing operations*. The skipper was required to abort the fishing event altogether if attempts to release encircled dolphins failed.

In addition to the abovementioned changes in fisher behaviour, two gear modifications were also included in the CoP. Firstly, a dolphin gate was added to the purse-seine net, which comprised a removable section of corkline along the top of the net (Fig. 2). When removed, the unsupported section of net sank, creating an opening for encircled dolphins to exit. Secondly, all vessels were required to carry purpose-built attachable weights to sink the corkline.

*

The fishery was reopened in late September 2005, with all fishing operations subject to the newly developed CoP. A second observer program was then conducted to assess the effectiveness of the CoP in mitigating operational interactions with dolphins.

### *4.3.5 Aims of this study*

The aims of this study were to:

1. Compare rates of encirclement and mortality of dolphins reported by fishers and recorded by independent observers.
2. Estimate the number of dolphins encircled and killed during each of the two study periods.
3. Measure the effectiveness of the CoP in reducing operational interactions of the SASF with dolphins.

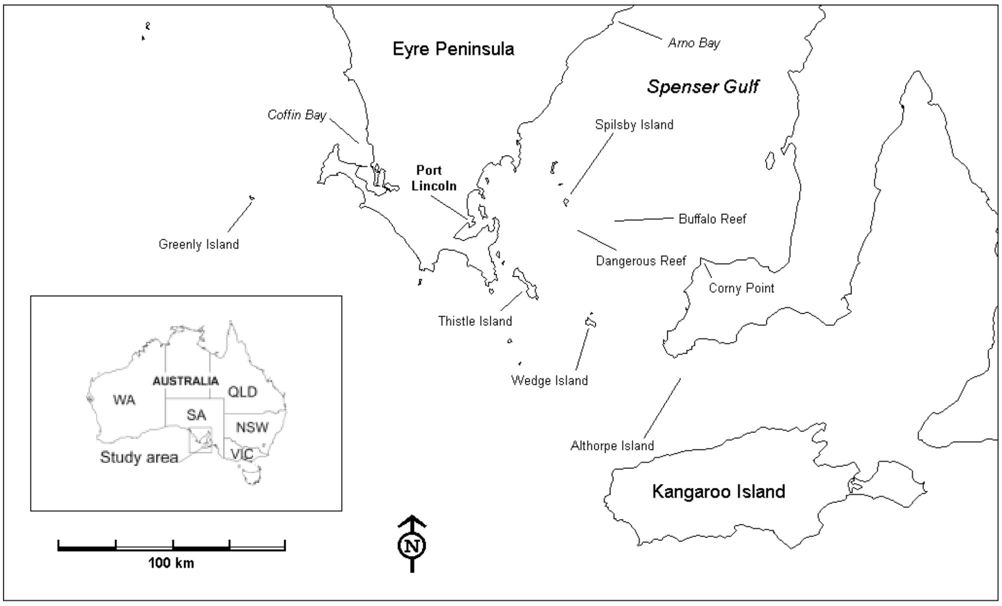



Figure 1 Location of the study area and of important sites.

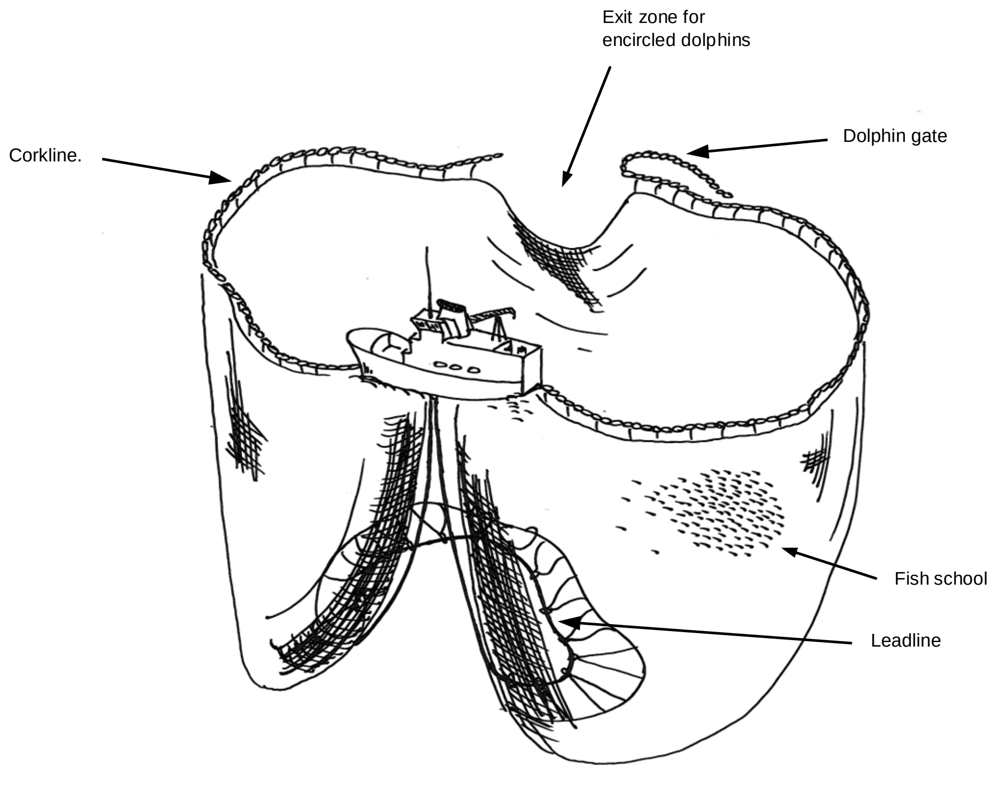



Figure 2 Schematic diagram of a typical purse-seine net, depicting the dolphin gate and the net folds beneath the vessel that form during pursing.

## 4.4 METHODS

### *4.4.1 Historical logbook data*

It has been mandatory for the fishery to lodge data relating to operational interactions with dolphins since April 1999. This information was obtained from SARDI to determine the level of fishing effort, plus the number and rate of operational interactions over that time. In addition, the monthly variation in fishing effort was calculated and compared with the incidence of operational interactions with dolphins, for the period between April 1999 and May 2004. The percentage of encircled animals that subsequently died was calculated and regressions analysis was used to determine if there was a relationship between the time of year and the incidence of encirclement.

### *4.4.2 Assessing the effect of introducing the CoP*

Two observer programs were conducted, one before and one after the introduction of the CoP, during November 2004–June 2005 and November 2005–June 2006, respectively. The two programs were conducted in the same months to reduce seasonal effects on sampling outcomes. Logbook data for these periods were collated and summarised.

During each of the two observer programs, one or more vessels carried one independent observer per trip upon the request of SARDI. Observations were made from a high, unobstructed vantage point such as the wheelhouse, wheelhouse roof or the bow, depending on the vessel and the prevailing weather conditions, and were concentrated within the corkline (Fig. 2). As all fishing occurred at night, observations were carried out either with the naked eye, assisted with binoculars (Gerber_ DLX/R 10 · 50) during moonlit periods, or with a night vision monocular (ITT_ N160) during periods of reduced visibility. As it was unlikely dolphins encircled at the beginning of the net-set could escape the pursed net without human assistance, it was unlikely encirclements or mortalities would not be detected, thus avoiding the chance of underestimation.

The date, location (latitude and longitude), total number of individual encirclements and mortalities and the number of encirclement and mortality events were recorded for each observed net-set. These data were used to determine temporal and spatial trends, plus the rates of operational interactions. Other details about the nature of the interactions were recorded, including: (i) the stage of the operation (net deployment, pursing, hauling and pumping) during which encircled dolphins were first detected; (ii) the time taken for crews to respond when encircled animals were detected; (iii) the nature and success of the release procedures; and (iv) swell height (to determine if it was related to the incidence of by-catch mortality).

In spite of low light conditions, encircled dolphins were typically detected early on during the fishing event. The behaviour of encircled dolphins was observed and categorised, using a combination of 'focal group sampling' (assessment of group behaviour) and 'predominant activity sampling' (the most frequent behaviour over a given sampling period; Altmann, 1974; Martin and Bateson, 1993; Mann, 1999). This was done to determine if behavioural cues indicating imminent death due to capture myopathy could be identified (Coe and Stuntz, 1980).

### *4.4.3 Data analysis*

The spatial and temporal distributions of fishing effort (the number of net-sets) were calculated from data obtained from each observer program and from concurrent logbook data. Regression analysis was used to determine the degree to which encirclement and mortality rates were correlated with the spatial and temporal distribution of fishing effort. For spatial data, the regression was based on the level of fishing effort and number of encirclements in each ten-by-ten kilometre grid square.

The effectiveness of the CoP was determined by comparing the mean encirclement and mortality rate before and after its introduction. To test the significance of change after implementing the CoP, a 1-tailed t-test was applied, because a reduction in encirclement and mortality rates were expected. Although observer and logbook data approximated a Poisson distribution, the t-test used is robust, provided the pooled sample size is greater than 40 (Moore and McCabe, 2003). This assumption was met in this study, with 49 observations made prior to the CoP being introduced and 89 observations after. The t-test for the null hypothesis of no difference in the mean rates of encirclements and mortalities of sets observed pre-CoP compared with those observed post-CoP was calculated (Rice, 1995). The variance of data collected pre-CoP was larger than during the post-CoP period, so was dealt with by pooling them and using the approximate formula for degrees of freedom (Rice, 1995; Moore and McCabe, 2003).

Power analysis was used to estimate the number of observations (ie. the number of net-sets monitored by observers) required to detect future changes in the encirclement and mortality rates, based on data obtained during the second observer program (post-CoP). The power to detect rate increases or decreases depended on the sample variance, sample size, the magnitude of the change that occurred and the degree of statistical significance of the change. Standard levels of significance ($\alpha$ = 0.05) and power ($\varphi$ = 80%) were used for these calculations. The sample size required to achieve this power and significance was calculated for prescribed levels of change in either encirclement or mortality rates: $\Delta_{\mathrm{p}} = (\mu_Y - \mu_X)/\mu_X$.

Power was written as a probability integral for the null hypothesis of no change in the t-distribution over tested levels of change in the observed rates $(\bar{Y} - \bar{X})$, from $(\bar{Y} - \bar{X})_{\text{crit}}$ to infinity, in order to determine if significant increases in either encirclements or mortalities had occurred. Thus, the probability that a future t-test with the same sample variance would yield a significant difference was calculated from:

$$\text{Power} = \int_{(\bar{Y}-\bar{X})=(\bar{Y}-\bar{X})_{\text{crit}}}^{+\infty} f\left[t = \frac{(\bar{Y}-\bar{X}) - \Delta_p \cdot \bar{X}}{\sqrt{\left(\frac{(n_X-1)s_X^2+(n_Y-1)s_Y^2}{n_X+n_Y-2}\right)\cdot\left(\frac{1}{n_X}+\frac{1}{n_Y}\right)}}, \text{df} = n_X + n_Y - 2\right] d(\bar{Y}-\bar{X})$$

where f(t, df) = probability density function of the t-distribution, with df degrees of freedom.

Assumed levels of change in encirclements and mortalities were calculated using power analysis and plotted as: (i) decreases of 10–90% (in increments of 10%); and (ii) increases of 100–700% (in increments of 100%). A 1-tailed t-test was applied in the power analysis because it was used to detect increases and decreases separately, relative to post-CoP encirclement and mortality rates.

## 4.5 RESULTS

### *4.5.1 Historical logbook data (1999-2004)*

Logbook data for the SASF between April 1999 and May 2004 indicated that operational interactions with dolphins were minimal, but variable between years. From the 3915 net-sets conducted over the five-year period, fishers reported 69 encircled dolphins and one death (Table 1). Encirclement and mortality rates reported were 0.0176 and 0.0003 dolphins per net set, respectively. According to the logbooks, only 1.5% of the encircled dolphins subsequently died. The number of encircled dolphins was strongly and positively associated with the location of reported fishing effort (Encirclement = 0:01 x location of effort + 0:01; $P < 0{:}01$; $R^2 = 0{:}50$), with most occurring in areas of high fishing effort in southern Spencer Gulf. No interactions were recorded along the north coast of Kangaroo Island and west of Coffin Bay, where the water was deeper than 60 m. There was no relationship between the number of encirclements recorded in each year and the corresponding level of fishing effort ($P = 0.77$; $R^2 = 0.06$). However, there was a significant, positive relationship between the number of encirclements recorded and monthly fishing effort (Encirclement = 0:01 x monthly effort + 2:68; $P = 0{:}02$; $R^2 = 0{:}44$). Fishing occurred in all months and encirclements occurred in each of them except November, although more occurred between April and June, when fishing effort was greatest. The only reported mortality occurred in April 2002.

### *4.5.2 Before introduction of the CoP (2004-2005)*

#### *4.5.2.1 Initial observer program*

The initial observer program was conducted between November 2004 and June 2005. A total of 87 encircled dolphins and 19 deaths of short-beaked common dolphins were recorded during 18 by-catch events, from 49 net-sets monitored over 89 nights (Table 1). The overall encirclement and mortality rates were 1.7755 and 0.3878 dolphins per net-set, respectively. A total of 21.3% of all encircled dolphins died. Given that 973 net-sets were recorded in fishery logbooks across the fishery over the same time period, 5.0% of all net-sets were monitored. Dolphins were observed bow riding and feeding on sardine schools prior to 81.6% (40 of the 49) of net-sets monitored. Assuming encirclements and mortalities occurred at the same rate across the remainder of the fishery over the same period, the estimated number of encirclements and mortalities was 1728 and 377, respectively.

Eight of the 11 vessels operating during this period carried an observer when requested by SARDI. Fishing activity was monitored throughout most of the area historically fished within the southern Spencer Gulf (Fig. 3a). Fishing did not occur near Coffin Bay and only a relatively small amount of effort was undertaken near Althorpe Island. The number of encirclements recorded by observers was strongly and positively associated with the location of fishing effort (Encirclement = 2:73 x location of effort + 0:03; $P < 0:01$; $R^2 = 0:79$), with most encirclements occurring east of Thistle Island and northeast of Wedge Island. No interactions occurred along the north coast of Kangaroo Island.

The number of interactions with dolphins varied between months, with most occurring in January and May (Fig. 4a). No interactions were recorded in November and December 2004, but low numbers of mortalities occurred between February and June 2005. The greatest numbers of encirclements occurred in January and May 2005. There was no relationship between the number of dolphin encirclements and monthly fishing effort ($P = 0.30$; $R^2 = 0.18$).

Seventy nine of the 87 encircled dolphins were initially observed alive. Most (62) of these were first detected soon after hauling had begun, once the deck lights were turned on, although some were detected earlier during pursing (14). Some (three) encircled dolphins were not detected until the net was brought alongside the vessel, prior to commencing pumping, although this only occurred during rough weather conditions.

Eleven of the 19 dolphins that died were initially observed alive, swimming at the surface, within the corkline. The average time taken for crews to respond to the presence of encircled animals and to initiate a release procedure was 135.9 ± 23.7 min on occasions when one of the 11 mortalities occurred, compared with 62.5 ± 6.8 min when encircled dolphins were released successfully (Fig. 5a). The remaining eight dolphins that died were already dead when they were first sighted and were detected within five minutes from the start of hauling, once the deck lights were turned on.

Although fishing generally occurred in good weather when swell height was typically 1 m or less, these mortalities occurred when the swell height was above 2.5m (Fig. 6). It is likely these eight animals became entangled in sub-surface net folds directly under the vessel during pursing and subsequently drowned (Fig. 2). In contrast, encircled animals that were initially observed alive but then later died occurred across all swell heights (Fig. 6).

Consistent behavioural patterns were observed in the 79 encircled dolphins that were initially observed alive and swimming freely inside the corkline (excluding the eight that were already dead), during the 18 encirclement events. Of particular note was the behaviour classified as ‘erratic swimming’, which provided the first clear indication that an encircled dolphin was becoming stressed. This behaviour was typified by frequent bursts of rapid swimming in no particular direction, with numerous bouts of tail fluke slapping on the surface of the water. Soon after this initial stress behaviour was observed, some individuals stopped swimming and became motionless in the water in a ‘vertical floating’ position, with beak, head and blowhole above the waterline. All of these animals exhibited ‘passive sinking’ soon after, whereby they began to float belly-up and then sink beneath the surface.

This sequence of behaviours was typically associated with imminent mortality, because the animals displaying them did not return to the surface of their own accord and subsequently drowned. Divers attempted to assist animals that exhibited these behaviours on a number of occasions by physically moving them toward the surface, but without success. The duration of the encirclement and the area within the corkline appeared to be associated with the behavioural sequence described above, although the two were probably confounded, making it difficult to determine the individual effect of each on dolphin behaviour.

During the initial observer program, fishing operations were not delayed or relocated on any occasion when dolphins were observed near a target sardine school. ‘No action’ was the most prevalent response when dolphins were detected prior to commencing fishing and during encirclements (Table 2). Only 15.6% of encircled dolphins escaped form the net without action being taken, although these almost always occurred when the corkline was pulled below the surface by an excessively large school of sardines exerting downward pressure. Other actions taken in order to release encircled dolphins were: opening the dolphin gate (Fig. 1), submerging the corkline by using weights, opening the front of the net, physical removal and aborting the net-set (Table 2). Interestingly, the gear modifications (ie. dolphin gate and weights) did not appear to be reliable tools for releasing dolphins, while opening the net front and aborting the net-set were very successful (Table 2).

#### *4.5.2.2 Logbook data: during initial observer program*

Fishery logbook data was collected from all active vessels over the same period as the initial observer program and they reported 63 dolphin encirclements and seven mortalities, from 973 net-sets (Table 1). The encirclement and mortality rates were 0.0648 and 0.0072 dolphins per net-set, respectively. From these figures, it is estimated they represent only 3.6% of the encirclement rate and 1.9% of the mortality rate recorded by independent observers during the initial observer program.

The number of encirclements recorded in logbooks was positively associated with the spatial distribution of fishing effort (Encirclement = 0:031 x location of effort + 0:012; $P < 0:01$; $R^2$ = 0:22), as was the case with the observer data. Encirclements occurred east of Thistle and Wedge Islands, east of Dangerous Reef and southeast of Althorpe Island. Mortalities were reported from west of Corny Point, between Thistle and Wedge Islands and east of Buffalo Reef. Although there was no temporal relationship with the recorded number of dolphin encirclements found in the observer data, a weak relationship was calculated from SASF logbooks (Encirclement = 0:035 x effort + 2:624; $P = 0:04$; $R^2 = 0:11$). The greatest number of encirclements occurred in January 2005, as was the case in the logbook data (Fig. 4b).

### *4.5.3 After introduction of the CoP (2005-2006)*

#### *4.5.3.1 Second observer program*

The second observer program was conducted between November 2005 and June 2006. Once again, the short-beaked common dolphin was the only dolphin species involved in operational interactions with the fishery. After the introduction of the CoP, 20 dolphins were encircled and 1 mortality was recorded from 89 monitored net-sets (Table 1). This equates to a significant reduction in the observed rate of encirclement by 87.3% to 0.2247 per net-set ($F = 5.36$; $df = 2$; $P < 0.01$) and mortality by 97.1% to 0.0112 per net-set ($F = 5.82$; $df = 2$; $P < 0.01$). The number of encircled animals that subsequently died after becoming caught in the net reduced from 21.8%

to 5.0%. A total of 753 net-sets were recorded in SASF logbooks across the fishery over the same time period, indicating 11.8% of net-sets were monitored. Therefore, an estimated 169 encirclements and 8 mortalities occurred across the entire fishery and over the same period, assuming the rates were constant.

All 12 vessels operating in the fishery during the second observer program, which included the eight that participated in the initial observer program, carried an observer at least once during the second observer program. Observations were concentrated in the southern Spencer Gulf region, although some fishing was monitored adjacent to Greenly Island, Coffin Bay and along south-western coast of the Eyre Peninsula (Fig. 3b). Encirclements predominantly occurred in areas of high fishing effort to the east of Thistle Island, Wedge Island and Dangerous Reef, and southeast of Althorpe Island. There were no interactions along the north coast of Kangaroo Island. The only mortality occurred near Althorpe Island in November 2005, soon after the program commenced (Fig. 4c).

The behaviours of encircled dolphins during the second observer program were similar to that described during the initial observer program. Encircled dolphins were detected earlier after the CoP was introduced due to crewmembers being assigned to searching for them, resulting in some encircled animals being detected before the deck lights were turned on and no animals being first detected during pumping. The average response time of fishers to encirclement events during the second observer program reduced by approximately 88.0% to 16.3 ± 4.4 min, compared with the initial observer program (Fig. 5b). All encircled dolphins that were initially observed alive and swimming at the surface within the corkline were successfully released. The only mortality that occurred was first detected dead soon after hauling had commenced. This death was attributed to drowning by entanglement in sub-surface net folds directly beneath the vessel (Fig. 2).

*

During the second observer program, fishing operations were delayed or relocated every time a dolphin was observed near the target school. No dolphin encirclements occurred when this avoidance guideline was followed (Table 2). Nonetheless, some dolphins were still encircled, because they were not detected prior to commencing fishing operations, meaning that delay and relocation strategies were not carried out. On these occasions, release procedures were used much more often than during the initial program, although the levels of success were similar in both programs (Table 2).

#### *4.5.3.2 Logbook data: during second observer program*

Logbook data collected during the second observer program indicated that 98 dolphins were encircled and five were killed from 753 net-sets across the fishery (Table 1). The encirclement and mortality rates were 0.1302 and of 0.0066 dolphins per net-set, respectively. Agreement between industry and observer data increased after the introduction of the CoP, with the encirclement and mortality rates recorded in logbooks increasing to 57.9% and 58.9%, respectively, of those recorded during the second observer program.

Encirclements occurred mainly in areas of high fishing effort, northeast of Thistle and Wedge Islands, east of Dangerous Reef and southeast of Althorpe Island and occurred in each month for the duration of the study period, with most occurring between February and March 2006 (Fig. 4d). Mortalities occurred northwest and west of Thistle Island and near Althorpe Island and were also temporally spread throughout the study period.

### *4.5.4 The power of future observer programs to detect changes in interaction rates*

At the standard levels of power ($\varphi$ = 80%) and significance ($\alpha$ = 0.05), it would not be possible to detect declines in the encirclement or mortality rates beyond those recorded in the second observer program, due to the low levels of interactions recorded following the introduction of the CoP (Fig. 7). Conversely, a tripling (200% increase) in the encirclement rate could be detected from as few as 21 observed net-sets, but 310 net-sets would be needed to detect a doubling (100% increase) in the encirclement rate. Similarly, a fivefold (400%) increase in the mortality rate could be detected if 57 net-sets were observed and a quadrupling (300% increase) could be detected if 198 net-sets were observed.

**Table 1 – Summary of short-beaked common dolphin encirclements and mortalities derived from observer records and industry logbook reports, presented as the total number of individuals, the number of events in which they occurred, plus the corresponding rate and estimate**

| Period | Net-sets | Operational interactions | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Encirclements | | | | Mortalities | | | |
| | | # Of animals | # Of events | Calculated rate (per-net-set) | Fleet wide estimate | # Of animals | # Of events | Calculated rate (per-net-set) | Fleet wide estimate |
| *Observer* | | | | | | | | | |
| Before CoP | 49 | 87 | 18 | 1.7755 ± 0.4882 | 1728 | 19 | 11 | 0.3878 ± 0.1265 | 377 |
| After CoP | 89 | 20 | 9 | 0.2247 ± 0.0795 | 169 | 1 | 1 | 0.0112 ± 0.0112 | 8 |
| *Logbook* | | | | | | | | | |
| 1999–04 | 3915 | 69 | 30 | 0.0176 | | 1 | 1 | 0.0003 | |
| Before CoP | 973 | 63 | 28 | 0.0648 ± 0.0079 | | 7 | 6 | 0.0072 ± 0.0027 | |
| After CoP | 753 | 98 | 47 | 0.1302 ± 0.0123 | | 5 | 5 | 0.0066 ± 0.0032 | |

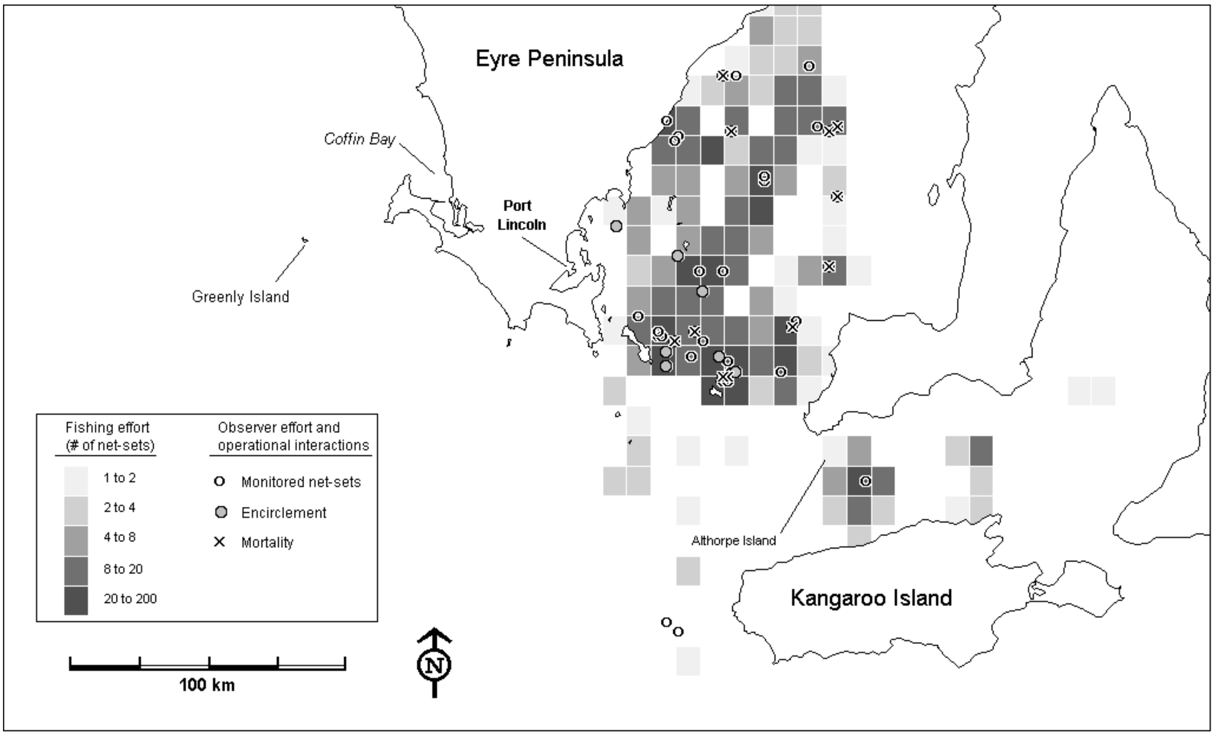
Values are derived from: historical records (1999–2004), plus data collected before (2004–2005) and after (2005–2006) the introduction of the CoP.

**Table 2 – Summary of the success of avoidance and release procedures used by fishers to mitigate dolphin by-catch**

| Guideline | CoP assessment | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Before CoP | | | After CoP | | | Combined | | |
| | # Records | # Successful | % Successful | # Records | # Successful | % Successful | # Records | # Successful | % Successful |
| *Avoidance measures* | | | | | | | | | |
| Delay/relocate | 0 | – | – | 15 | 15 | 100.0 | 15 | 15 | 100.0 |
| *Release procedures* | | | | | | | | | |
| No action | 32 | 5 | 15.6 | 0 | – | – | 32 | 5 | 15.6 |
| Attach corkline weights | | 8 | 53.3 | 4 | 2 | 50.0 | 19 | 10 | 52.6 |
| Open dolphin gate | 2 | 1 | 50.0 | 7 | 3 | 42.9 | 9 | 4 | 44.4 |
| Open net front | 18 | 14 | 77.8 | 7 | 6 | 85.7 | 25 | 20 | 80.0 |
| Physical removal | 18 | 16 | 88.9 | 3 | 3 | 100.0 | 21 | 19 | 90.1 |
| Abort net-set | 2 | 2 | 100.0 | 6 | 6 | 100.0 | 8 | 8 | 100.0 |

Values are derived from: historical records (1999–2004), plus data collected before (2004–2005) and after (2005–2006) the introduction of the CoP.

**a** Before CoP introduced




**b** After CoP introduced

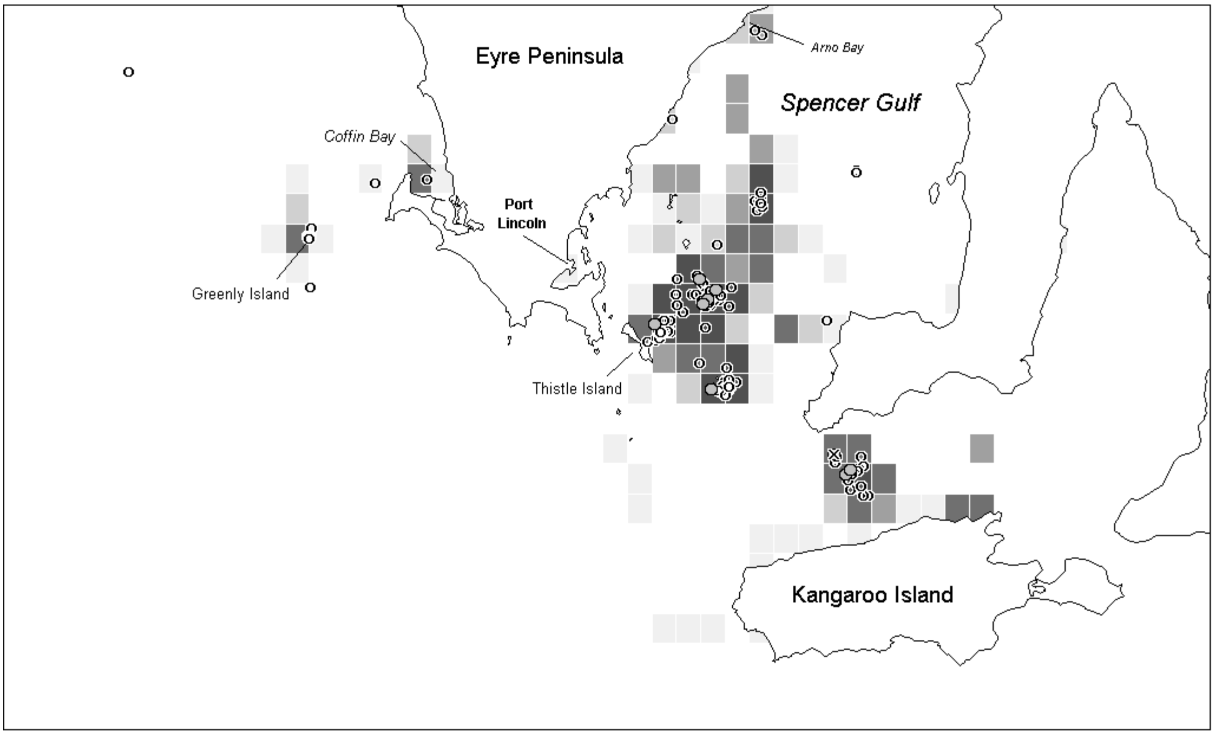



Figure 3 Spatial distribution of fishing effort and dolphin encirclements and mortalities, before (a) and after (b) the introduction of a CoP.

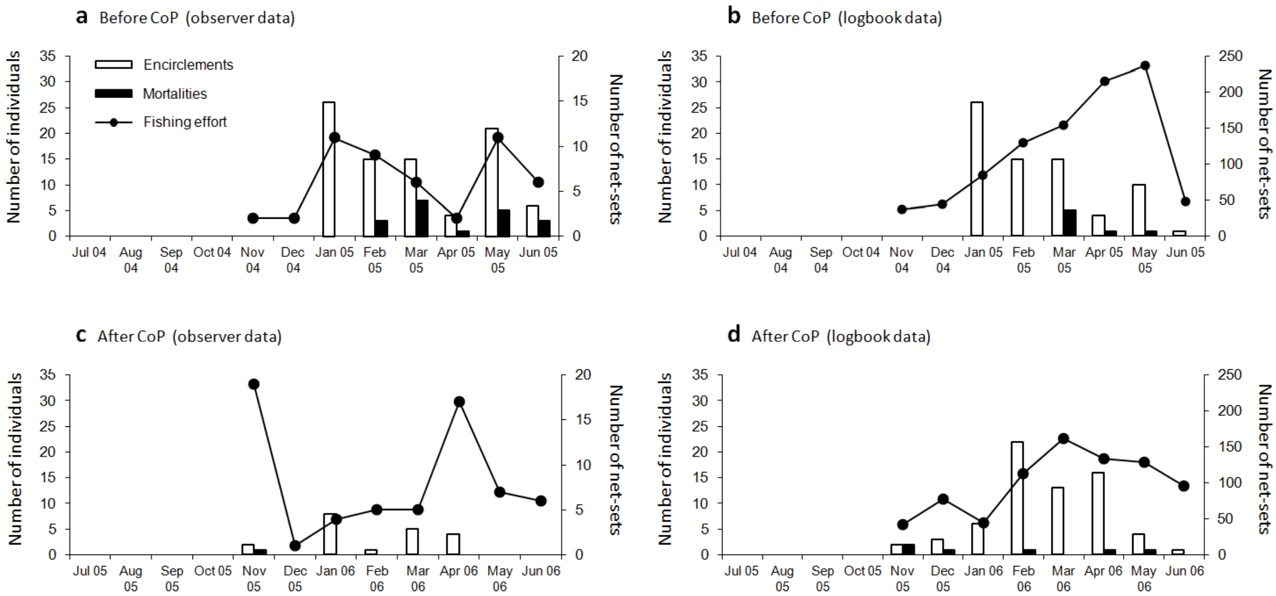



Figure 4 Intra-annual (monthly) patterns in fishing effort, plus the number of dolphin encirclements and mortalities, before (a,b) and after (c,d) the introduction of a CoP.

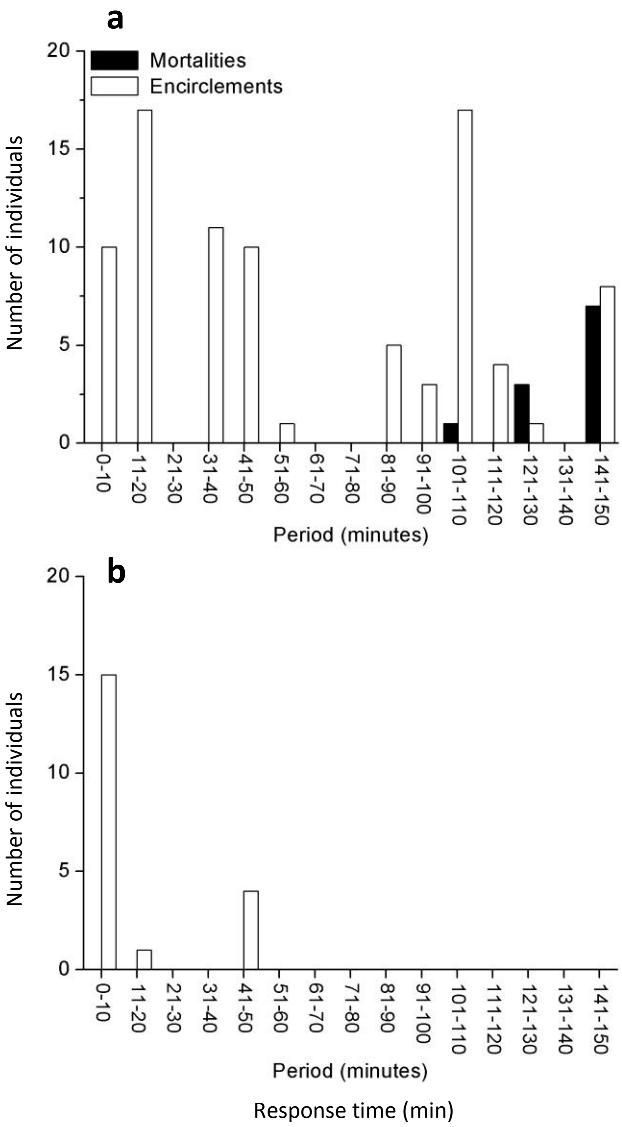



Figure 5 The times taken for crews to implement dolphin release procedures after detecting encircled dolphins, before (a) and after (b) the introduction of the CoP.

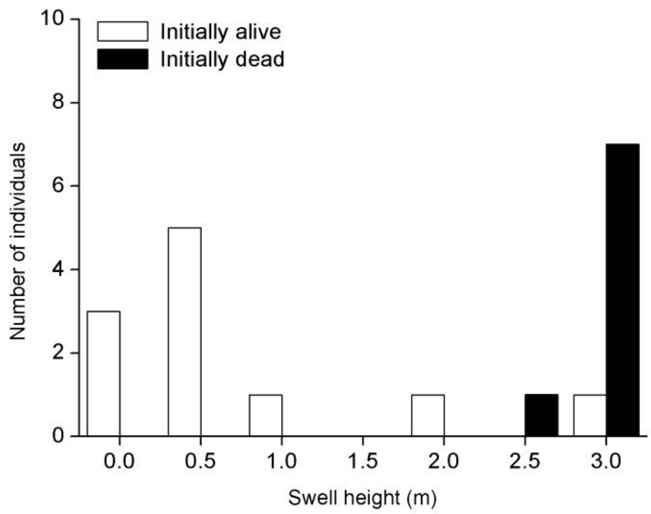



Figure 6 The relationship between swell height and dolphin mortalities, derived form data collected before the introduction of the CoP.

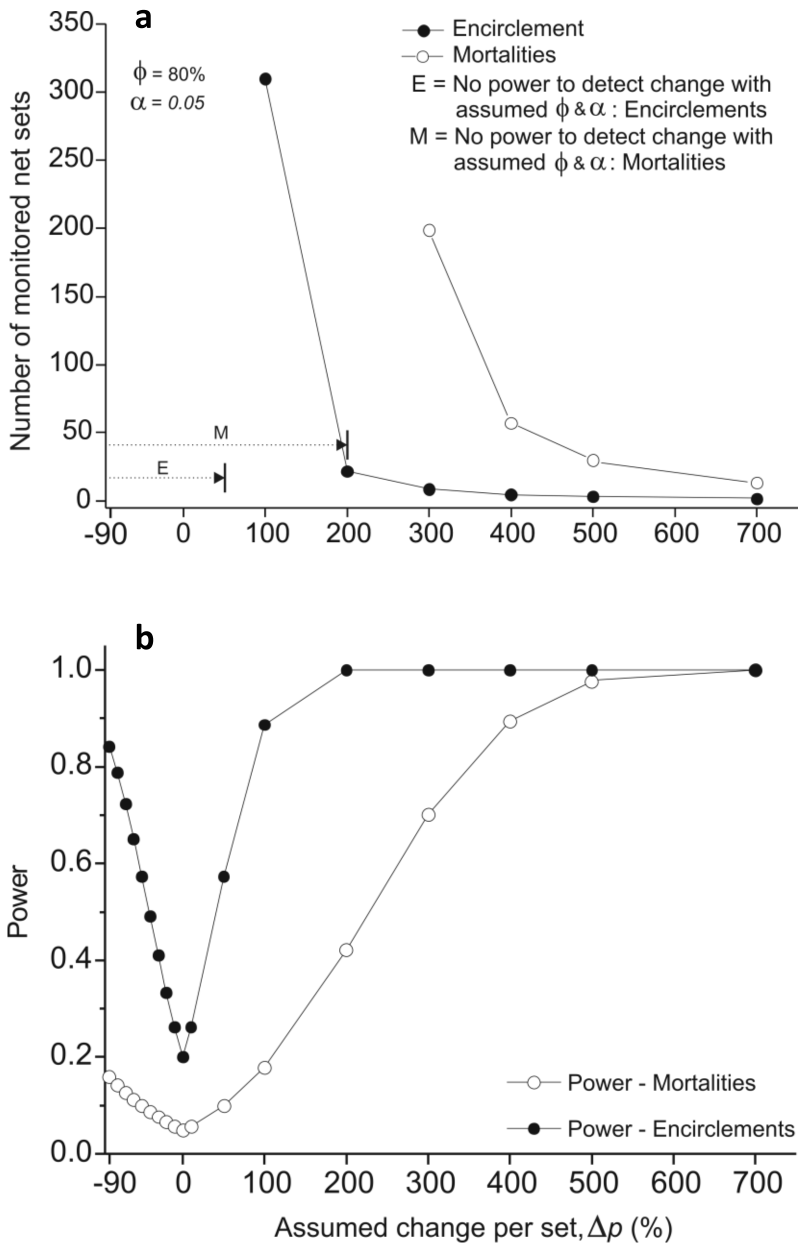



Figure 7 Results of analysis for the number of net-sets required to detect a prescribed level of change in encirclements and mortalities (a), plus the corresponding level significance and power (b).

## 4.6 DISCUSSION

### *4.6.1 The CoP as the preferred dolphin by-catch mitigation tool*

Logbook and observer data suggest operational interactions with dolphins occur across the geographic range of the SASF, with spatial patterns of encirclement strongly associated with the level of fishing effort. Although there were areas where operational interactions did not occur, the majority occurred in fishing hotspots, suggesting dolphins were attracted either by the aggregation of large schools of sardines, or by the activity of the fishing vessels that converged upon them. In addition, historical and observer logbook records from the first observer program suggest there was a marked intra-annual correlation in the number of operational interactions between the sardine fishery and dolphins, with most encirclements occurring when fishing effort was greatest. Once again, this suggests the dolphins are attracted either directly by the fish aggregations, or the fishing effort by proxy.

The fact this pattern did not exist during the second observer program is likely the result of increased efforts by fishers to prevent encirclements and mortalities, rather than a departure from this behaviour by the dolphins. Historical data indicate these results are unlikely to be confounded by the movement of dolphins in and out of the fishing grounds, because encirclements occurred the year round. This suggests either the possibility of a resident population (although its size and range remain unknown), or year round visitation by a larger and more transient population. As such, the incidence of encirclements of dolphins by the SASF follows seasonal fluctuations in fishing effort, rather than intra-annual variations in the numbers of dolphins in the fishing grounds. Therefore, spatial and temporal closures would not be suitable for mitigating operational interactions of common dolphins with this fishery, thus justifying the introduction of a CoP focused on modifying fisher behaviour and fishing gear.

### 4.6.2 Success of the CoP at mitigating dolphin by-catch

The high rates of operational interactions with short-beaked common dolphins recorded during the initial observer program were of the same magnitude as those reported in the developmental pilchard fishery in Queensland, although mortality rates in this study were almost three times as high (Hale et al., 1999; Staunton-Smith and Ward, 2000). The CoP that was subsequently introduced to the fishery was similar to that proposed for the Queensland fishery and resulted in large reductions in the observed rates of encirclements (87.3%) and mortalities (97.1%). By-catch estimates for the entire fleet during each of the two seven month observer programs suggest the number of encirclements declined from 1728 to 169 and the number of mortalities declined from 377 to 8, after the introduction of the CoP. These results demonstrate that changes in fisher behaviour and fishing gear modification can mitigate the impacts of commercial fisheries on marine mammals. Similar changes to fisher behaviour and fishing gear resulted in comparable reductions in dolphin by-catch in the eastern tropical Pacific tuna purse-seine fleet (Gosliner, 1999).

A marked cultural change occurred in the fishery during this study, with fishers becoming more aware of their need to mitigate the impacts of their activities on dolphins. A similar evolution was reported as the principal driving force behind the reduction in dolphin by-catches by the US tuna fleet in the ETP (Gosliner, 1999). The improvements in SASF operations are due in part to the philosophy of inclusion of all stakeholders in the development of the CoP, plus the adoption of realistic changes to fishing practices that could be thoroughly and rapidly implemented. During the second observer program in particular, it was mandatory under the CoP for at least one member of the crew to actively search for dolphins prior to deploying the net and for fishing operations to be delayed or relocated if a dolphin was observed near the target school.

These guidelines are likely to have been responsible for the significant reduction in encirclement rates subsequently recorded.

It also became mandatory under the guidelines of the CoP for fishers to continue searching for encircled dolphins during the entire fishing event and to implement release procedures immediately upon detecting dolphins inside the net. As a result, encircled dolphins were more likely to be detected earlier and this was reflected in the marked reduction in response times by 76.9% to about 16 min. This guideline helped to ensure that encircled dolphins were released before they began to display behaviours commonly associated with mortality events.

Stress behaviours typically occurred immediately before a mortality event, after a considerable amount of time had elapsed. Although not quantified, the eleven mortalities that occurred under these circumstances also took place at a time when the circumference of the net had diminished considerably, suggesting the elapsed time and the area within the pursed net were confounded, making it impossible to distinguish their individual effect. Nonetheless, the CoP could be modified to encourage fishers to release dolphins as soon as they are detected, but no later than when one of these behaviours is observed. This would abolish the need for data relating to the response time of fishers and the space available within the net. The association of stress behaviours with mortality events was first described in the ETP tuna purse-seine fishery (Norris et al., 1978; Coe and Stuntz 1980; Gosliner, 1999). The suitability of using behavioural indicators of stress was a key element of the proposed response to the encirclement of dolphins in the southern Queensland pilchard fishery (Staunton-Smith and Ward, 2000). A detailed investigation of the behaviour of encircled dolphins would assist in further refining the CoP and mitigating dolphin mortalities in this and similar fisheries.

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An important part of the CoP was that fishers were required to abort the net-set if all other attempts to release encircled dolphins were unsuccessful. During the initial observer program, some fishers were reluctant to abort fishing operations to release encircled dolphins, which lead to protracted response times and subsequent mortalities. However, after the introduction of the CoP, fishers aborted the net-set if other attempts to release encircled dolphins were unsuccessful and this change was associated with the marked reduction in dolphin mortality rates. It is also notable that fishers became better at interpreting dolphin behaviour after the introduction of the CoP and that some fishers aborted net-sets as soon as rafting behaviour was observed, or sooner.

The gear modifications trialled during the CoP were surprisingly unsuccessful when compared with other alternatives. This result was surprising given the apparent success of analogous apparatus in the ETP (Gosliner, 1999). One explanation for this difference between the two fisheries may be subtle differences in the installation and use of these devices. Some observers in the sardine fishery commented that the weight of the sinking net tended to draw the corkline together and close the dolphin gate, thus preventing the escape of dolphins. These gear modifications will need to be refined if they are to become an effective tool for releasing encircled dolphins in the SASF.

### *4.6.3 Improvements of fishery logbook reporting*

Several previous studies have suggested that fishery logbook data are unsuitable for measuring the number or rate of operational interactions with marine mammals (Bache, 2003; Romanov, 2002; Walsh et al., 2002; Baum et al., 2003; Dans et al., 2003). A similar conclusion could be drawn in this study from the comparison of logbook and observer data for the fishery both prior to and after the introduction of the CoP. Nonetheless, there was an increase in the level of agreement in encirclement and mortality rates sourced from logbooks during the second observer program when compared with observer data, rising from 3.6% to 57.9% and from 1.9% to 58.9%, respectively. However, the current level of reporting of dolphin by-catch by the SASF still requires improvement, due to the large proportion that remained unreported.

In addition to ongoing underreporting, fishers may have modified their behaviour in the presence of observers to reduce the probability of operational interactions with dolphins. The 'observer effect' was reported in the ETP tuna purse-seine fishery, where a significantly higher number of dolphins were killed on vessels carrying an observer monitoring the compliance of dolphin release procedures than in the presence of observer specifically monitoring the number of dolphins killed (Wahlen and Smith, 1985). While it would be impossible to quantify this categorically (ie. presence and absence of observers), this behaviour among fishers is likely to result in the observer data providing an underestimate of the actual numbers and rates of dolphin by-catch. This is not an uncommon problem when monitoring the ecological effects of fisheries, with recent analysis of fishery logbook data in a New Zealand fishery indicating it only reported about half of its by-catch when compared with observer data (Burns and Kerr, 2008).

#### *4.6.4 Power to detect change*

The low rates of encirclement and mortality that were achieved after the introduction of the CoP have implications for future monitoring. Power analyses showed that an observer program of the scale conducted in this study (i.e. 100–200 monitored net-sets per year) would not have the capacity to detect further reductions in interaction rates. This presents a problem for measuring future proposed improvements to the CoP, but also indicates that observed interaction rates were at a low level during the second observer program. A similar situation occurred in the eastern tropical Pacific tuna fishery, where the large reductions in dolphin mortality made it difficult to assess the effectiveness of further improvements in fisher behaviour and fishing gear (Gosliner, 1999). The ability to detect increases in interaction rates is also related to the level of observer coverage. A total of 89 net-sets were monitored during the second observer program, although the power analyses indicate that approximately 310 observed net-sets would be needed to detect a 100% increase (doubling) in the encirclement rate, while 198 observed net-sets would be required to detect a 300% (fourfold) increase in the mortality rate. Therefore, under the 11.8% observer coverage achieved during the second observer program, only large increases in interaction rates could be detected, by which time major departures from the CoP are likely to have occurred.

### *4.6.5 Potential impacts on the short-beaked common dolphin population in SA*

In general, very little is known about the potential impacts of fishery induced by-catch mortalities on common dolphin populations. Prior to this study, the extent and nature of their operational interactions with commercial fisheries had only been investigated in another similar fishery on Australian northeast coast (Hale et al. , 1999). In the absence of reliable population estimates for common dolphins in Australian waters, it is impossible to determine if they are risk of decline.

In spite of these uncertainties, some life history parameters provide insights into the potential impacts of by-catch mortality, especially if the population is already small. Females typically become sexually mature at between 7.9 and 9.5 years of age and live for up to 25 years (Danil and Chivers, 2007; Westgate and Read, 2007). Gestation lasts for between 11 and 12 months, a calf is produced every 2.1 years and they exhibit a fecundity rate of between 25% and 33%, resulting in the production of 7–8 calves in their lifetime (Danil and Chivers, 2007; Westgate and Read, 2007). These figures suggest the reproductive capacity of common dolphins is very low.

However, they are a best estimate of production, because they do not account for calves that do not reach sexual maturity due to disease and predation, nor those that are killed or orphaned by fishing activities (Archer et al., 2001, 2004; Noren and Edwards, 2007). Therefore, the removal of even low numbers of animals from a population may have large and deleterious impacts.

In Australia, the little amount of research that has been conducted has focused on diet and population genetics. Common dolphin carcasses collected in South Australia revealed they ate squid (two species) and at least seven families of teleost fish (at least 16 species; Kemper and Gibbs, 2001). These findings suggesting they are opportunistic foragers that commonly feed on small pelagic schooling fishes, including sardines, which explains their frequent encounters with SASF fishing activity. Although a highly mobile and apparently ubiquitous species, a recent population genetics study demonstrated that animals from South Australia were genetically distinct from animals in Tasmanian waters, some 1400 km to the southeast (Bilgmann, 2007). This suggests a genetic boundary between the two populations and the subsequent limitations to immigration from adjacent populations. In contrast, very little genetic differentiation was found to exist between short-beaked common dolphin populations in the eastern tropical Pacific, northwest, northeast and southwest Atlantic, and the southwest Indian Oceans, which are separated by 4000–17,000 km (Natoli et al., 2006; Amaral et al., 2007).

In summary, there is little known about the status and size of the common dolphin population in South Australian waters. Notwithstanding, their limited reproductive capacity and the apparent restrictions to immigration in the South Australian population suggest the population is vulnerable to adverse impacts under relatively low levels of fishery induced by-catch mortality.

### 4.6.6 Recommendations

Given the limited understanding of the impacts of the bycatch mortality sustained by the South Australian short-beaked common dolphin population in recent times, there is a need to obtain information on the abundance and boundaries of the population. Tools that could assist in obtaining such information include population surveys and population genetic studies, respectively. This information would improve our understanding of the effect operational interactions have on their populations.

Further refinement of the CoP should include a requirement for fishers to monitor the behaviour of encircled dolphins, not just their presence or absence. As such, fishers would need to become familiar with behaviours associated with stress and imminent mortality. These behaviours could then be used to help categorically identify when a net-set should be aborted, in preference to the more complicated and time consuming approach which involves removing encircled animals while saving the catch. Even though the latter is the preferred option for the fisher who is following an economic imperative, this study has shown that encircled dolphins are at greater risk of dying when stress behaviours are observed. In addition, the current gear modifications should be reviewed in light of their poor performance and consideration given to alternative strategies, including opening the front of the net, which has already been employed with a high degree of success by some fishers.

The apparent association between swell height and mortality events warrants further investigation also, especially as eight of the animals that died probably becoming caught in net folds under the vessel and died before reaching the surface. The CoP was not applicable in these cases, because these animals were dead when first observed. These incidents only occurred when the swell height was above 2.5 m, indicating the need to include additional guidelines in the CoP that address this potential cause of dolphin mortality. Finally, even though the discrepancy between the rates of operational interactions recorded by fishers and by observers diminished in the second observer program, there is a need to address continued underreporting by fishers. Fishers may be encouraged to improve their reporting with the introduction of tougher penalties, or increased observer coverage, both of which cold result in a financial burden to individual fishers and the SASF in general. Until this is achieved, partial or full self management of marine mammal by-catch by the SASF should not be considered.

### *4.6.7 Conclusions*

Our results indicate that CoPs can be useful tools for managing and mitigating operational interactions between short-beaked common dolphins and purse-seiners in the SASF and may be applicable elsewhere under similar circumstances. The CoP significantly reduced the fishery's operational interactions with dolphins, with the avoidance and release strategies likely responsible for reducing the number of encirclements and mortalities. In contrast, the gear modifications outlined in the CoP appeared to have little effect. In general though, the CoP as it currently stands has met its four underpinning principals, suggesting it should remain as the tool for mitigating dolphin by-catch. Nonetheless, when considering the shortfalls to the CoP that were highlighted during this study, it would benefit from being refined to include clear references to: (i) when and how fishers should search for dolphins; (ii) the kind of behaviours that indicate stress in encircled animals; (iii) the most effective methods for avoiding and releasing them; and (iv) weather conditions, such as swell height. Such improvements to the CoP for dolphin by-catch mitigation, in addition to ongoing observer coverage and improved logbook reporting, could result in the SASF becoming a 'best practice' example of this important issue.

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## Assessing the effectiveness of the Great Australian Bight Marine Park in protecting the endangered Australian sea lion *Neophoca cinerea* from bycatch mortality in shark gillnets

*They write pieces they do not much enjoy writing, for papers they totally despise, and the sad process ends by ruining their style and disintegrating their personality, two developments which in a writer cannot be separated, since his personality and style must progress or deteriorate together, like a married couple in a country where death is the only permissible divorce.*

Claud Cockburn

## Statement of Authorship

Title of publication: Assessing the effectiveness of the Great Australian Bight Marine Park in protecting the endangered Australian sea lion *Neophoca cinerea* from bycatch mortality in shark gillnets

Publication status: Published, September 2011

Publication details: *Endangered Species Research* 14, 203-216

**Author Contributions**

By signing the Statement of Authorship, each author certifies that their stated contribution to the publication is accurate and that permission is granted for the publication to be included in the candidate's thesis.

Principal: Derek J. Hamer

Contribution: • Established industry relationships • Developed the project • Conducted background research • Developed experimental and data collection protocols • Supervised fieldwork at Bunda Cliffs from which spatial data for Australian sea lion at-sea movements were collected • Undertook all gill-net vessel monitoring • Collated, analysed and presented data • Wrote entire manuscript.

Signature: Date: 24 October 2012

Co-Author: Tim M. Ward

Contribution: • Made available funds to assist in part funding of salary • Reviewed drafts of manuscript, providing particular advice on structure and presentation.

Signature: Date:

Co-Author: Peter D. Shaughnessy

Contribution: • Raised awareness about the problem of Australian sea lions becoming by-catch in demersal gill-nets • Reviewed drafts of manuscript.

Signature: Date: 28 October 2012

Co-Author: Simon R. Clark

Contribution: • Initiated project as the Great Australian Bight Marine Park Manager and provided funding and opportunity to undertake the project • Provided established links to a variety of stakeholders and participants • Assisted with fieldwork at Bunda Cliffs.

Signature: Date: 28 October 2012

## 5.1 ABSTRACT

The Endangered Australian sea lion *Neophoca cinerea* occurs in low numbers, exhibits low fecundity, extreme philopatry and substantial population genetic structure at the breeding colony level. These traits may increase susceptibility to population decline, with additional mortality as bycatch in shark gillnets being a possible threat. The Great Australian Bight Marine Park (GABMP) was established, in part, to protect the small and remote Bunda Cliffs population from anthropogenic impacts such as commercial fishing. This study investigated the effectiveness of the GABMP in reducing spatial overlap between Australian sea lions and gillnets and in preventing bycatch. An independent fishery observer program reported a mortality rate of 0.0206 individuals $km^{-1}$ of gillnet set within the GABMP, amounting to between 4 and 15 (confidence bounds of standard error of the estimate) individuals killed there during the most recent breeding cycle. A mortality rate of 0.0093 individuals $km^{-1}$ of gillnet set was recorded across the broader GAB region, amounting to between 14 and 33 individuals killed each breeding cycle during recent times, and between 128 and 177 over the 10 yr since the GABMP was established in the mid-1990s. These reported bycatch levels are unlikely to be sustainable and may represent minimum estimates, because drowned individuals may drop out of the gillnet and go unobserved. A tracking program involving 9 females (5.6% of the estimated female population) demonstrated that they spent only 27.7% of their time inside the GABMP. Four of them regularly travelled more than 180 km from home, or 9 times further than the southern boundary of the GABMP. These results indicate that the level of protection afforded by the GABMP to Australian sea lions residing at Bunda Cliffs is unlikely to reduce bycatch to below the levels that would reduce the risk of decline in this small population. Suggested improvements to the GABMP include a year-round closure to gillnetting, low bycatch limits and extension of the southern boundary further south. Additional regulatory mechanisms may be needed in the gillnet fishery to minimise its impact on this and other small Australian sea lion populations.

## 5.2 ACKNOWLEDGEMENTS

We sincerely thank the owners of The Fish Factory Pty. Ltd. (L., K. and P. Toumazos) and their skippers (M. Larsson FV 'Lutarna' and G. McDonald FV 'Jean Bryant') for access to vessels and enthusiastic support during the observer program; A. Ivey (South Australian Research and Development Institute, SARDI) for assistance with collecting observer data; D. Frazer (Environmental Biodiversity Services, EBS) and B. Page (SARDI) for assistance with sea lion captures; P. Codrington, P. Bache and D. Skinner (State Emergency Service, SES) for facilitating vertical access to the base of Bunda Cliffs; T. Timmis and M. Daniels (Australian Fisheries Management Authority, AFMA) for providing historical fishery effort data; S. Childerhouse (Australian Marine Mammal Centre, AMMC) for reviewing several drafts; M. Sumner (University of Tasmania) for advice on analysis of spatial foraging data; S. Candy and J. McKinlay (Australian Antarctic Division, AAD) for advice on analysis of bycatch data; J. Fitzpatrick (AAD) for assistance with editing figures. We also gratefully acknowledge the funding support of the GABMP Steering Committee. Approval to conduct these activities was granted by the University of Adelaide Animal Ethics Committee (S-008-2007) and conducted under an SA Department of Environment and Heritage (DEH) permit (A-24684).

## 5.3 INTRODUCTION

Although commercial sealing is now banned in most regions, many pinniped populations were decimated or extirpated during the 18th, 19th and 20th centuries and are yet to recover to pre-exploitation levels (Taylor 1982, Wickens et al. 1991, Ling 1999, Harwood 2001, David & van Sittert 2008). In more recent times, fishing effort has intensified to meet the increasing demand for fish products for human consumption, thus leading to increased incidence of direct or operational interactions between pinnipeds and commercial fisheries. As such, the outcomes of these interactions are now considered to be a primary contemporary threat to the conservation of many pinniped populations (Woodley & Lavigne 1991, Wickens 1995). These events may result in individuals becoming entangled in portions of lost fishing gear, which can lead to injury, starvation and eventual death (Fowler et al. 1990, Page et al. 2004). Individuals may also become entangled in active fishing gear, thus leading to a more immediate death by drowning (Beverton 1985, Bonner 1989, Woodley & Lavigne 1991, Wickens 1995, Shaughnessy et al. 2003).

Such incidences are known to occur in Australian waters. For example, Australian fur seals *Arctocephalus pusillus doriferus* enter the cod end of trawl nets to depredate caught fish off the west coast of Tasmania and sometimes drown if they fail to successfully exit the large structure (Hamer & Goldsworthy 2006). Similarly, young Australian sea lions *Neophoca cinerea* depredate bait and catch from rock lobster traps in Western Australian (WA) coastal waters and sometimes drown if they become caught in the small entrance (Campbell et al. 2008b). Australian sea lions also depredate small sharks caught in demersal gillnets off South Australia (SA) and sometimes drown when they become entangled in the fine meshes (Shaughnessy et al. 2003, Page et al. 2004).

### 5.3.1 *Australian sea lion life history, status & vulnerability*

Unlike Australian fur seals, Australian sea lion populations have failed to recover from exploitation and recolonise their former geographic range (Warneke 1982, Gales et al. 1994, Ling 1999, Goldsworthy et al. 2003, 2009b, Shaughnessy et al. 2003, 2005, 2006, Campbell et al. 2008a, Robinson et al. 2008). Their peculiar life history may exacerbate the effect of fishery related losses. Firstly, the 17.4 to 17.8 mo long breeding cycle of the Australian sea lion has resulted in a lower level of fecundity (lifelong reproductive output) when compared with annually breeding pinnipeds, such as the conspecific New Zealand fur seal *Arctocephalus forsteri* (Higgins 1993, Gales & Costa 1997). This reduces the capacity of, and rate at which, a population can recover from declines. Secondly, pronounced genetic structure is exhibited between sexually mature females from different breeding sites even in situations where geographic overlap in foraging range is extensive, thus inferring that many breeding colonies are distinct populations (Campbell et al. 2008a, Goldsworthy & Lowther 2010). Thirdly, this situation may have arisen due to the prevalence of philopatry among sexually mature females, where individuals almost exclusively give birth and breed at their natal colony (Gales & Costa 1997). As such, suitable but unoccupied sites adjacent to current breeding colonies are unlikely to host founder populations. Fourthly, annual estimated pup production is low at 2441 to 3610 (9300 to 17 364 individuals overall) and 62% of the 76 known populations in SA and WA yield fewer than 25 pups (Gales et al. 1994, Goldsworthy et al. 2009b, Shaughnessy et al. 2011; see our Fig. 1). As such, the probability of population decline and extinction at Australian sea lion breeding sites is comparatively greater than for most other pinniped species, even if just a few individuals are lost each breeding cycle due to operational interactions with fishing gear.

### *5.3.2 Extent, nature & impact of operational interactions*

The demersal shark gillnet fishery, which is managed by the Australian Fisheries Management Authority (AFMA), has operated in Australia's southern continental shelf waters since the early 1970s and has changed little since that time (Walker et al. 2005). The fishery targets gummy sharks *Mustelus antarcticus* and school sharks *Galeorhinus galeus* using monofilament polyamide and polypropylene gillnetting, hung between a weighted foot rope that holds it stationary on the benthos and a floated headline that holds it upright in the water column. Waters around Australia fall into 2 zones: those belonging to each state government (inside 5.56 km from the coastline) and those belonging to the Australian Government (between 5.56 and 370.4 km from the coastline). Gillnets set inside SA (state government) waters cannot exceed 1.8 km in length pursuant to the SA Fisheries Management Act 2007, and those set in Australian Government waters cannot exceed 4.2 km in length pursuant to the Australian Government Fisheries Management Act 1991. Over 17 000 km of gillnet were set annually between 2000 and 2008 adjacent to the SA coastline, and effort was distributed across waters shallower than 183 m in depth, which comprises most of the shelf area (Goldsworthy et al. 2010, Hamer et al. 2010).

Little is known about the foraging habits of Australian sea lions. An early study of free-ranging and captive Australian sea lions concluded that their diet was difficult to determine from scats, because they rarely contained identifiable hard prey remains (Gales & Cheal 1992). However, freshly regurgitated material and the stomach contents of deceased Australian sea lions revealed that benthic dwelling fish, cephalopods, crustaceans and sharks were consumed (McIntosh et al. 2006). Recent advances in biochemical studies, such as polymerase chain reaction techniques and fatty acid analysis, have further confirmed the presence of benthic prey in the Australian sea lion diet (Peters et al. 2007, Baylis et al. 2009). This is reinforced by studies of dive behaviour of sexually mature females from 4 breeding sites, which indicate that foraging individuals typically spend most of their time at or near the benthos (Costa & Gales 2003, Fowler et al. 2006, Goldsworthy et al. 2009a). Recent findings indicate that Australian sea lions forage across a large proportion of the shelf waters adjacent to SA (Goldsworthy et al. 2010, Hamer et al. 2010).

Shark gillnetters and Australian sea lions are likely to concentrate their efforts in similar locations at sea, which may result in deleterious impacts on the latter. Of all entanglements recorded at the breeding colony at Seal Bay (Kangaroo Island, SA), 55% involved monofilament gillnets (Page et al. 2004). One demersal gillnetter estimated that 20 Australian sea lions were incidentally caught and drowned in their gillnets annually during the 1990s adjacent to the SA coastline (Shaughnessy et al. 2003). Another estimated that 12 were killed in their gear over a recent 12 mo period (Adelaide Now 2011). Other entangled individuals may initially survive, only to die later from associated injuries or starvation (Fowler 1987, Fowler et al. 1990, Page et al. 2004). A high level of shark gillnetting occurs adjacent to the Seal Bay breeding colony (Goldsworthy et al. 2010, Hamer et al. 2010), whose population declined by 1.1% each breeding cycle between 1985 and 2002/03 (Shaughnessy et al. 2006). Interestingly, the Dangerous Reef (Spencer Gulf, SA) breeding colony increased by 0.6% each breeding cycle between 1975 and 2000/01 and then by 4.8% each breeding cycle between 2000/01 and 2006/07 (modified from Goldsworthy et al. 2007), after gillnetting was banned there (SA Government Gazette, 22 March 2001, p. 1060–1061; SA Government Gazette, 2 May 2001, p. 1703). These findings suggest that entanglement and bycatch related mortalities of Australian sea lions in demersal gillnets may be causing population declines at some breeding colonies and may be suppressing recovery at others.

### *5.3.3 Mitigating impact through statutory protection*

Prior to 2000, before the proclamation of the Australian Government Environment Protection Biodiversity Conservation Act 1999 (EPBC Act), Australian fisheries were not required to report operational interactions with marine mammals. Between 2000 and 2007, only 10 seals of unidentified species were reported in the demersal gillnet fishery, suggesting that many events remained undetected or unreported (Hamer 2007). Growing concern about the conservation of the Australian sea lion resulted in the Australian Government Department of Environment (Department of Sustainability, Environment, Water, Population and Communities; DSEWPaC), the administrator of the EPBC Act, obtaining advice that identified entanglement and bycatch mortality in demersal gillnets as a major threatening process to the species. Consequently, the Australian sea lion was listed as vulnerable under the EPBC Act in 2005 (DEWHA 2008) and as Endangered on the International Union for the Conservation of Nature Red List in 2008 (Goldsworthy & Gales 2008). Pursuant the EPBC Act, an ecological assessment of major commercial fisheries must be undertaken every 5 yr. Upon receipt of the ecological assessment for the demersal gillnet fishery, DSEWPaC recommended that AFMA and the fishery (1) mitigate the number of bycatch mortalities and (2) improve reporting procedures.

### ***5.3.4 Australian sea lions & the Great Australian Bight Marine Park***

During the early 1990s, a comprehensive survey identified 10 sites where Australian sea lions breed on a narrow boulder field at the base of Bunda Cliffs, at the head of the Great Australian Bight (GAB) in SA (Dennis & Shaughnessy 1996, Shaughnessy et al. 2005; Fig. 1). These sites are inaccessible other than by abseiling. An estimated 161 pups were recorded from ground counts, equating to between an estimated 613 and 774 animals, or between 4.8 and 9.3% of the species (Dennis & Shaughnessy 1996). Unfortunately, the data obtained from surveys since that time have involved remote cliff-top surveys with binoculars, suggesting that an unknown quantity of pups that shelter under the boulders would remain uncounted (South Australia Department for Environment and Heritage, unpublished data). In addition, the data are of poor quality because brown pups, moulted pups and juveniles were on occasion incorrectly classified by unskilled observers, and surveys were often incomplete and poorly timed. As such, subsequent survey data cannot be used to construct a reliable population trend. Bunda Cliffs is geographically isolated from the nearest populations to the west by about 650 km (Recherche Archipelago) and to the east by 300 km (Nuyts Archipelago). The philopatry and population genetic structure observed in other regions (Campbell et al. 2008a, Goldsworthy & Lowther 2010) and the smaller distances that sexually mature females are known to travel from their natal colony (Fowler et al. 2007) suggest that Australian sea lions at Bunda Cliffs comprise 1 or more isolated populations.

The Great Australian Bight Marine Park (GABMP) was established in the mid-1990s pursuant to the Australian Government EPBC Act and the SA Government National Parks and Wildlife Act 1972, in part to protect Australian sea lions that reside and forage within its boundaries (Edyvane 1998, NHT 2005). The GABMP is situated in the northern reaches of the GAB waters and includes the area adjacent to the Bunda Cliffs. The GABMP straddles SA and Australian Government waters, comprising a Sanctuary Zone (SZ) and a Conservation Zone (CZ) located in SA waters and a Marine Mammal Protection Zone (MMPZ) in adjacent Australian Government waters (DEH 2005, NHT 2005). The SA and Australian Government components have their own management plan, although the GABMP is effectively managed as 1 entity under a bilateral agreement. The GABMP covers an area of approximately 21 500 km$^2$, and its boundaries are delimited by the WA border in the west (129° 00’ E), Cape Adieu (132° 00’ E) in the east, the SA coastline in the north and the 31° 47’ S latitude line in the south (Fig. 1).

Gillnet fishing is prohibited throughout the year only in a small portion of the GABMP; all of the SZ and a small part of the CZ in the west. The boundary of the permanently closed area extends from the low water mark at the SA-WA border, then due south 9.3 km, then east 9.3 km at 9.3 from the coastline, then a further 290 km east at 5.6 km from the coastline to Cape Adieu (Edyvane 1998, NHT 2005, Gibson 2008; our Fig. 1). Gillnet fishing in the remainder of the

GABMP, which extends approximately 21 km south of the low water mark at its farthest point, is prohibited for 6 mo of each year, between 1 May and 31 October (DEHAA 1999, DEH 2005; our Fig. 1). Despite these areas of protection and the distances they extend offshore, sexually mature females tracked from Seal Bay travelled up to approximately 75 km offshore (Fowler et al. 2006). Therefore, individuals residing at Bunda Cliffs may still be at risk of becoming bycatch in demersal gillnets, when foraging (1) in parts of the GABMP where fishing is permitted (i.e. the CZ and MMPZ) and (2) outside the GABMP altogether.

### *5.3.5 Need & aims*

The use of MPAs to protect endangered pinniped species has proven successful in the past, by excluding fishing in areas to mitigate the chance of incidental bycatch mortality (e.g. Hawaiian monk seal *Monachus schauinslandi*: Lavigne 1999; Mediterranean monk seal *M. monachus*: Pires et al. 2008; New Zealand sea lion *Phocarctos hookeri*: Wilkinson et al. 2003). Given that Australian sea lion populations are likely to be vulnerable to even low levels of additional mortality and that shark gillnetting is permitted to occur within a substantial portion of the GABMP for half of the year, it would be useful to assess whether it provides effective protection to resident Australian sea lions. To achieve this, this study aims to determine:

(1) The level of bycatch mortality in shark gillnets specifically in the GABMP when it is open to fishing and generally across the GAB region;

(2) At-sea movements of females residing at Bunda Cliffs; and

(3) The trend in gillnet fishing effort in the GAB region over the last decade since the GABMP was established.

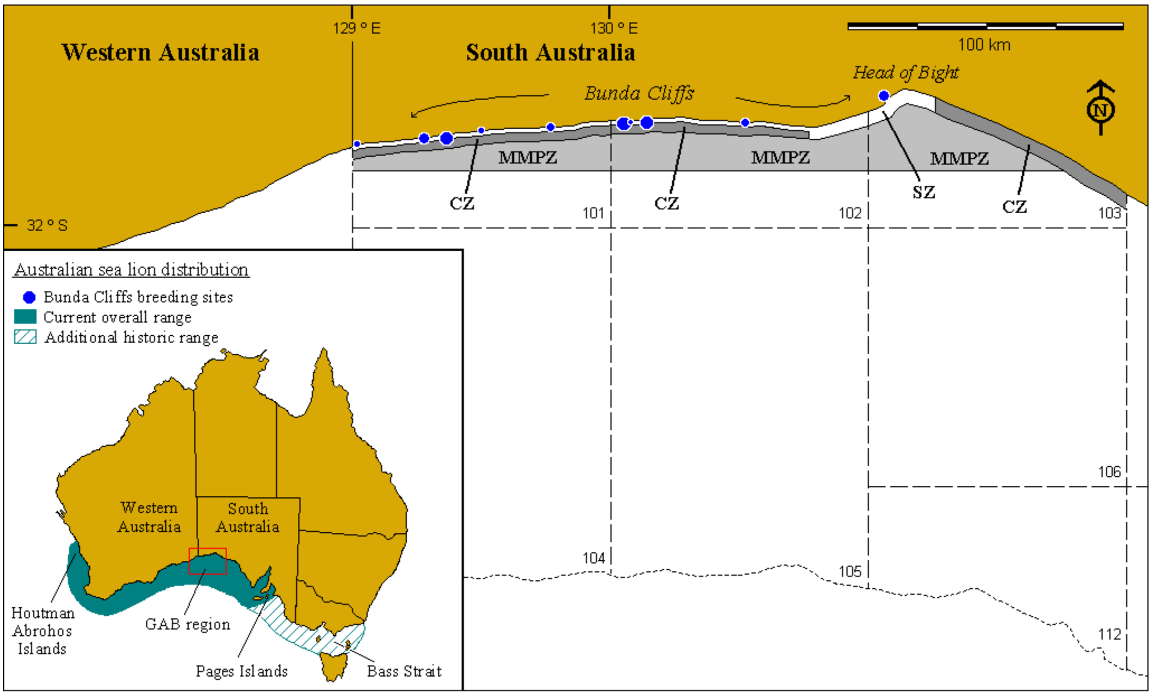


Figure 1 Great Australian Bight Marine Park (GABMP) comprising the Sanctuary Zone (SZ), Conservation Zone (CZ) and Marine Mammal Protection Zone (MMPZ). Australian Fishery Management Authority (AFMA) fishery management areas (MFAs) 101 to 106 and 112 are also presented and were used to define the GAB region for the purposes of t his study. The southern limit of the MFAs is the 183 m (100 fathom) depth line (short dashed line). The 10 Australian sea lion breeding sites along Bunda Cliffs and the overall range of the species in south-western Australia are also shown.

## 5.4 MATERIALS & METHODS

### *5.4.1 Observed bycatch mortalities & fishing effort*

An independent observer opportunistically accompanied 1 of 2 shark gillnet vessels fishing in the GAB region over the 2 yr study period, between January 2006 and December 2007. The GAB region was defined as the area within AFMA-derived Marine Fishery Areas (MFAs) 101 to 106 and 112, from 129° to 132° E and from 31° 30’ to 33° 30’ S, within which the GABMP is situated (Fig. 1). The principal task for the observer was to record the incidence and location of all fishing events and of Australian sea lions entangled and drowned in the gillnet. Where possible, age class and sex were also recorded. Observations were made during hauling by leaning outboard of the gunwale, in order to obtain an unimpeded view of the gillnet ascending through the upper several metres of the water column to the surface, then onto the net roller.

### *5.4.2 Fishery-wide gillnetting effort*

Partially summarized shark gillnet fishing effort data derived from fishery logbooks were obtained from AFMA. Since 1 July 2007, it has been mandatory for all vessels in the shark gillnet fishery to record the exact location of fishing events by degrees and minutes of latitude and longitude. Prior to this, fishers were only required to record the number of the MFA they fished in. MFAs conform to degrees of latitude and longitude, thus measuring 92 to 95 km along parallels of latitude at 31° to 34° S and about 111 km along meridians of longitude, or between 10 212 km2 and 10 545 km2 in area, respectively. The decision was made to use MFA-scale effort data, because they were consistently collected over the study period, whereas point location data were not.

*

The effort data were used to calculate the number of kilometres of gillnet set in each year and within each MFA and within each of the 3 areas within the GABMP, namely the SZ, CZ and MMPZ. It was assumed, based on discussions with fishers, that maximum permitted net lengths were used, i.e. 1.8 km in SA waters and 4.2 km in Australian Government waters. Given that it was not possible to determine the exact location of fishing events when recorded at the MFA scale, effort in MFAs intersected by the SA–Australian Government boundary and by the GABMP boundaries, namely 101, 102 and 103, was apportioned according to area.

### *5.4.3 Bycatch rates & estimates*

The number of observed bycatch mortalities was recorded and the bycatch rate calculated by dividing this figure by the number of kilometres of gillnet observed hauled. This was done for 5 zones of interest: (1) SA waters in the GABMP (SZ and CZ), (2) Australian Government waters in the GABMP (MMPZ), (3) all of the waters in the GABMP (SZ, CZ and MMPZ combined), (4) the remaining waters of the GAB region as defined in this study (to the south of the GABMP) and (5) the entire GAB region. Estimates of bycatch were extrapolated for each of the 5 regions, by multiplying the 5 bycatch rates by the overall number of kilometres of gillnet set in each zone, as reported in AFMA fishery logbooks. This was done over 3 time periods: (1) the 2 yr study period, (2) a 17.6 mo (nominal) breeding cycle and (3) a 10 yr period that approximated the time from when the GABMP was proclaimed to the conclusion of this study period. In the absence of data, spatial and temporal distribution of Australian sea lion bycatch and shark gillnetting effort across the region was assumed to be even. Estimates of bycatch were rounded up to the nearest whole number to reflect whole individuals. Due to observer effort being less than 100%, the approximate 95% confidence interval (CI) for each bycatch mortality estimate was calculated using the standard error for each estimate (SEE) technique:

$$SEE = \sqrt{1 - \left(\frac{observer\ effort}{fishing\ effort}\right)} \cdot mortality\ estimate$$

where variance is scaled by 1 minus the sample fraction (i.e. the finite population correction) and assuming that each estimate was based on a Poisson distribution (Cochrane 1977, Zar 1999). The lower and upper confidence bounds (CBs) of the bycatch estimates were then calculated by subtraction and addition, respectively, of 2 times the SEE from the mortality estimate (Cochrane 1977, Zar 1999).

#### *5.4.4 At-sea movements of sexually mature females*

Satellite-linked platform transmission terminals (PTTs; KiwiSat 101, Sirtrack) were attached to sexually mature female Australian sea lions at the 2 largest breeding sites at Bunda Cliffs, referred to in this study as the western site (31° 38′ 35′′ S, 129° 22′ 59′′ E) and the central site (31° 35′ 20′′ S, 130° 03′ E). The estimated number of pups counted at these 2 sites in 1994 was 38 and 43, respectively (Dennis & Shaughnessy 1996). Lactating females were chosen, because they produce the next generation and were identified by the presence of a suckling pup prior to capture, or by the presence of milk upon capture.

Four PTTs were deployed at the western site after the end of a breeding season in April 2006, and 5 PTTs were deployed at the central colony during the following breeding season in May 2007. Access to the narrow and undulating terraces at the base of Bunda Cliffs, where the breeding sites were located, was hindered by a large boulder field and 100 m high vertical cliffs. The only means of access was via the cliffs, which required the use of specialised abseiling and climbing equipment. A purpose-built, conical net was used to capture and immobilise target animals. Anaesthesia was facilitated with 100% isoflorane (Isoflo™, Veterinary Companies of Australia), delivered via a gas anaesthesia apparatus fitted with a Cyprane Tec III vaporiser (Advanced Anaesthetic Specialists). The degree of induction was controlled by adjusting the concentration and delivery rate of Isoflo™, based on vital signs such as heart rate, rate and depth of breathing, degree of openness of the airway, gum colour, capillary return, and eye and tail reflexes. The PTTs were positioned alongside the mid-dorsal line, 10 cm posterior of the fore-flipper pits. Attachment was made to the guard hairs using Araldite® 2107 (Huntsman Advanced Materials).

Location data were obtained from Service Argos. Class 3, 2, 1 and 0 positions were used in the analysis, while A, B and Z class positions were omitted due to their inaccuracy (Sterling & Ream 2004, Costa et al. 2010). With the use of a saltwater conductivity switch, each PTT paused transmission of location data when the tracked individual hauled out for longer than 1 hour and immediately recommenced upon re-entering the water, thus allowing individual foraging trips to be identified prior to analysis. The removal of the data in between foraging trips eliminated a near-shore bias of foraging time in areas adjacent to the western and central sites.

The remaining location data were used to establish the spatial distribution of foraging effort, as a value of time, by redistributing the effort into 1 $km^2$ grid cells, using R (version 2.3.0, R Foundation for Statistical Computing) and the specifically written script *trip* (version 1.1, M.D. Sumner, University of Tasmania). Erroneous locations were identified by *trip* as those that exceeded a maximum possible pinniped linear swim speed of 2 $ms^{-1}$ (Kuhn et al. 2009) from either of the 2 adjacent locations and were subsequently redistributed closer to those 2 adjacent locations. The average linear swim speed between 2 recorded locations was then used to assign each 15 min parcel of time along that line to the appropriate grid cell, thus allowing the time spent in each cell to be calculated.

Location fixes at the beginning and end of each foraging trip were often inaccurate or absent, possibly due to Bunda Cliffs hindering satellite communication. As such, the exact time a tracked animal left or returned to the breeding site was often unclear. To address this, exact coordinates for the breeding site were manually included in the data set as the first and last position for each manually identified foraging trip. The script *trip* was used to determine the additional time taken to travel between the breeding site and the first or last recorded location for each trip, using the maximum linear swim speed of 2 $ms^{-1}$.

The spatial distribution of foraging effort, as time, was calculated for the 5 regions in the GAB and GABMP, as described above. To accurately reflect the contribution of foraging effort by each of the 2 breeding sites, resulting values for each grid cell were weighted according to the contribution of each tracked animal to the overall tracking time calculated for the breeding site it originated from and of each breeding site to the overall tracking time calculated for both breeding sites. The time spent in each grid cell or area was plotted using MapInfo Professional® and Vertical Mapper® (versions 8.0 and 2.5 respectively, MapInfo Corporation).

The mean maximum distance and mean direction from the breeding site were also calculated for each tracked animal. Mean direction was established by calculating the bearing from the breeding site to the maximum distance travelled during each foraging trip. Maximum distances less than 5 km from the breeding site were not included, to minimise directional errors associated with data obtained from Service Argos.

## 5.5 RESULTS

### *5.5.1 Observed bycatch mortalities & fishing effort*

Four Australian sea lion bycatch mortalities were observed during the study period (Table 1, Fig. 2). All dropped out of the gillnet and sank when it was hauled above the waterline at the end of the fishing event. Due to the distinctive pelage colouration and size of the individuals, the observer positively identified 2 as sexually mature females, one as a bull and the other as a juvenile of unknown sex. Three, excluding the bull, were caught within the GABMP, with 1 caught in the CZ just outside the AFMA closure approximately 3.8 km offshore and the other 2 caught in the MMPZ just outside the CZ. The bull was caught approximately 46 km south of the GABMP. Over the 2 yr study period, a total of 431.4 km of gillnet (113 fishing events) was observed hauled in the GAB region, of which 145.8 km (45 fishing events) occurred in the GABMP (Table 1, Fig. 2).

### *5.5.2 Fishery-wide gillnetting effort*

Between January 1998 and December 2007, 197 689 km of gillnet was set in SA, of which 16 375 km was set in the GAB region (AFMA unpublished data). During the 2 yr study period, 3292.8 km of gillnet was set in the GAB region, of which 538.1 km occurred in the GABMP and 2754.7 km occurred to the south, across the remainder of the GAB. Fishing effort in the GAB region declined by 73.8% between 1998 and 2002 (from 3001 to 786 km), then increased by 403.9% between 2002 and 2007 (from 786 to 3175 km; Fig. 3). The decline coincided with target stock depletion in the late 1990s and the introduction of quota in 2001, while the increase coincided with stock recovery and renewed interest by demersal shark gillnet fishers in the GAB region (Fig. 3). Given that fishers were only required to record fishing effort at the MFA scale prior to 1 July 2007 and voluntarily by degrees and minutes thereafter, the spatial distribution of fishing effort for most of the data collected during the study period was very coarse. As such, it was not possible to accurately measure changes in geographic distribution over time.

### *5.5.3 Bycatch rates & estimates*

Based on the observed bycatch and fishery data, the Australian sea lion bycatch mortality rate for the study period was 0.0093 ind. $km^{-1}$ of gillnet set across the entire GAB region, 0.0206 ind. $km^{-1}$ in the GABMP and 0.0035 ind. $km^{-1}$ to the south, across the remainder of the GAB (Table 1). In contrast, the mortality rate calculated from AFMA logbooks for all SA shelf waters for the period October 1999 to January 2006 (i.e. from when the EPBC Act took effect to the beginning of this study) was 4.9 × 10–5 ind. $km^{-1}$ of gillnet set, based on 7 reported mortalities and 143 752 km of gillnet set (AFMA unpublished data). Comparison of rates derived from observer data and AFMA data suggest that those reported in this study were 190, 420 and 71 times higher, respectively.

During the study period, the observer program monitored 13.1% of fishing effort across the GAB region, 27.1% in the GABMP and 10.4% to the south of the GABMP in the remainder of the GAB (Table 1). An extrapolated estimate of Australian sea lion bycatch mortality based on all gillnet fishing activity within the GABMP during the most recent breeding cycle was 9 ± 2.7 (SEE; CBs: 4,15) individuals. Extrapolated estimates of bycatch mortality across the broader GAB region, as defined in this study, amounted to 23 ± 4.6 (CBs: 14,33) killed there during the most recent breeding cycle, 31 ± 5.2 (CBs: 21,42) during the 2 yr study period and 152 ± 12.2 (CBs: 128,177) during the 10 yr period since the GABMP was proclaimed.

### *5.5.4 At-sea movements of sexually mature females*

The 4 Australian sea lions tracked from the western site were monitored for 971 d (mean ± SD = 243 ± 37 d) and the 5 tracked from the central site were monitored for 215 d (mean = 43 ± 13 d;

Table 2). These 9 individuals constituted 5.6% of sexually mature females, based on the 1994 estimate of 161 pups. The mean distance travelled from the western site was 83 ± 35 km (mean of maximum distances travelled: 180 km) and from the central site was 30 ± 18 km (mean maximum: 69 km; Table 2, Fig. 2). The mean and mean maximum distances travelled from the western site were 2.8 and 2.6 times greater than from the central site, respectively. Of all foraging trips, 96 ± 17% (maximum 100%) of those recorded from the western site and 67 ± 12% of those recorded from the central site went beyond the southern boundary of the GABMP. Individuals tracked from the western site travelled in a south-westerly direction (mean bearing = 205 ± 26°), while individuals from the central site travelled south (mean bearing = 187 ± 39°). There was no spatial overlap between animals tracked from the western and central sites, although 8 of the 9 animals swam distances farther from their natal colony than the 63 km separating the 2 sites.

Individuals tracked from the western site spent 24.1% of their time at sea within the GABMP (14.4% in the SZ and CZ and 9.7% in the MMPZ) and 75.9% of their time in adjacent Australian Government waters (45.4% off SA and 30.5% off WA; Table 3, Fig. 2). Four of the 5 animals from the western site were tracked for 7 to 9 mo, during which time little changed in the geographic distribution of consecutive foraging trips. Individuals tracked from the central site spent 53.4% of their time at sea in the GABMP (27.3% in the SZ and CZ and 26.1% in the MMPZ) and 46.6% of their time in adjacent Australian Government waters off SA (Table 3, Fig. 2).

Table 1. Summary of Australian sea lion by-catch in demersal shark gill-nets in the Great Australian Bight Marine Park (GABMP) and also across the entire Great Australian Bight (GAB) region as defined in this study.

| Management Zone | Observer data summary | | | By-catch estimates | | | | |
|---|---|---|---|---|---|---|---|---|
| | Jan 2006 – Dec 2007 [1] | | | Jan 2006 – Dec 2007 | | Recent [2] | Jan 1998 – Dec 2007 [3] | |
| | Observer effort [4] | Mortalities observed | Mortality rate [5] | Fishing effort | Mortality estimate [6] | Mortality estimate | Fishing effort | Mortality estimate |
| SZ and CZ (SA waters) | 32.4 | 1 | 0.0308 | 89.7 | 3±1.4 (1,6) | 3±1.5 (1,6) | 305.5 | 9±2.8 (4,15) |
| MMPZ (Comm. waters) | 113.4 | 2 | 0.0176 | 448.4 | 8±2.4 (4,13) | 6±2.2 (2,11) | 1528.2 | 27±5.0 (18,37) |
| GABMP (SZ, CZ and MMPZ) [7] | 145.8 | 3 | 0.0206 | 538.1 | 11±2.8 (6,17) | 9±2.7 (4,15) | 1833.7 | 38±5.9 (27,50) |
| Remaining GAB waters [8] | 285.6 | 1 | 0.0035 | 2754.7 | 10±3.0 (5,16) | 8±2.7 (3,14) | 14541.3 | 51±7.1 (37,66) |
| Entire GAB region | 431.4 | 4 | 0.0093 | 3292.8 | 31±5.2 (21,42) | 23±4.6 (14,33) | 16375.0 | 152±12.2 (128,177) |

Abbreviations: SZ – Sanctuary Zone; CZ – Conservation Zone; MMPZ – Marine Mammal Protection Zone.

1 The 24 month period during which the study was conducted;
2 A recent 17.6 month breeding cycle, in which the by-catch estimate is a fraction of the 24 month study period (i.e. 17.6/24 = 0.7333);
3 A 10-year period, approximating the time from when the GABMP was proclaimed to the time when the study was concluded;
4 All observer and fishing effort measured in kilometres;
5 Number of observed mortalities divided by the number of kilometres of gill-net observed hauled;
6 Mortality rate multiplied by fishing effort, with standard error of the estimate (SEE) and lower and upper confidence bounds (CBs) in parentheses, rounded up to nearest whole number for individuals;
7 Combines the area of the SZ, CZ and MMPZ;
8 Across the remainder of the GAB region as defined in this study, to the south of the GABMP.

Table 2. Summary of tracking data of 9 sexually mature female Australian sea lions tracked from 2 sites at Bunda Cliffs, South Australia.

| Deployment summary | | | Distance travelled (km) [2] | | Direction travelled (deg.) [2] |
|---|---|---|---|---|---|
| Seal I.D. | Observed | # of days [1] | Mean [3] ± SD | Maximum | Mean [3] ± SD |
| *Western site (B8): 2006* | | | | | |
| 55933 | with pup | 189 | 93 ± 41 | 187 | 228 ± 15 |
| 55962 | lactating | 273 | 65 ± 29 | 181 | 211 ± 22 |
| 55973 | suckling | 252 | 99 ± 29 | 193 | 205 ± 37 |
| 55975 | lactating | 257 | 74 ± 40 | 158 | 174 ± 29 |
| | MEAN | 243 | 83 ± 35 | 180 | 205 ± 26 |
| | TOTAL | 971 | | | |
| *Central site (B5):* | | | | | |
| 55938 | suckling | 49 | 26 ± 16 | 68 | 192 ± 36 |
| 55940 | lactating | 33 | 39 ± 16 | 63 | 168 ± 22 |
| 55964 | with pup | 36 | 16 ± 14 | 53 | 210 ± 44 |
| 55974 | lactating | 34 | 25 ± 20 | 66 | 172 ± 53 |
| 55978 | suckling | 63 | 44 ± 26 | 84 | 192 ± 42 |
| | MEAN | 43 | 30 ± 30 | 69 | 187 ± 39 |
| | TOTAL | 215 | | | |
| | OVERALL MEAN | 132 | 53 ± 27 | 117 | 195 ± 33 |
| | OVERALL TOTAL | 1186 | | | |

1 The satellite transmitter deployment period;
2 The distance of each location fix from the deployment site (either B8 or B5);
3 The mean of all trips from the deployment site.

Table 3. Summary of the proportion of spatial utilisation of, by management zone and number of hours, of nine sexually mature female Australian sea lions from two sites at Bunda Cliffs, South Australia.

| Management Zone | Time spent in each zone | | |
|---|---|---|---|
| | Western site (%) | Central site (%) | Combined (%) |
| SZ & CZ (SA waters) | 14.4 | 27.3 | 16.0 |
| MMPZ (Comm. waters) | 9.7 | 26.1 | 11.7 |
| GABMP (SZ, CZ & MMPZ) | 24.1 | 53.4 | 27.7 |
| Remaining GAB waters [1] | 45.4 | 46.6 | 45.5 |
| Western Australia [2] | 30.5 | 0.0 | 26.8 |
| TOTAL (%) | 100.0 | 100.0 | 100.0 |
| TOTAL (hours) | 16658.0 | 2331.6 | 18989.6 |

1 Across the remainder of the GAB region as defined in this study, to the south of the GABMP;
2 All waters to the west of the GABMP, adjacent to WA.

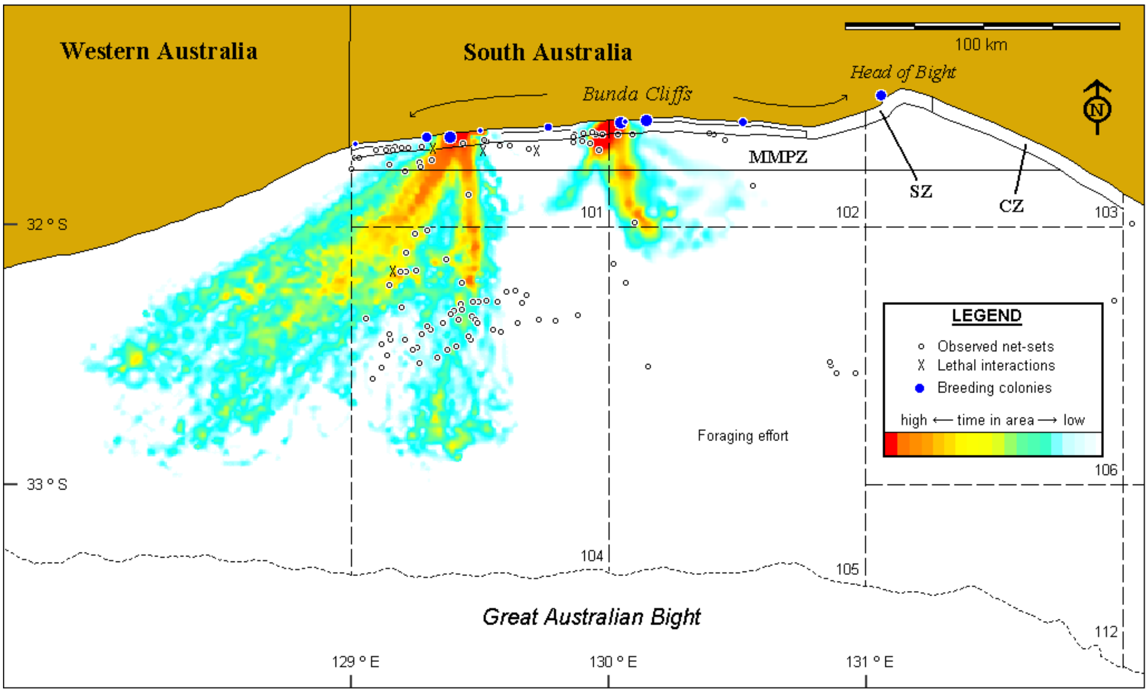


Figure 2 Summary of at-sea movements of 9 sexually mature female Australian sea lions tracked from 2 sites at Bunda Cliffs, South Australia. The location of observed gill-net sets and by-catch mortalities are also presented.

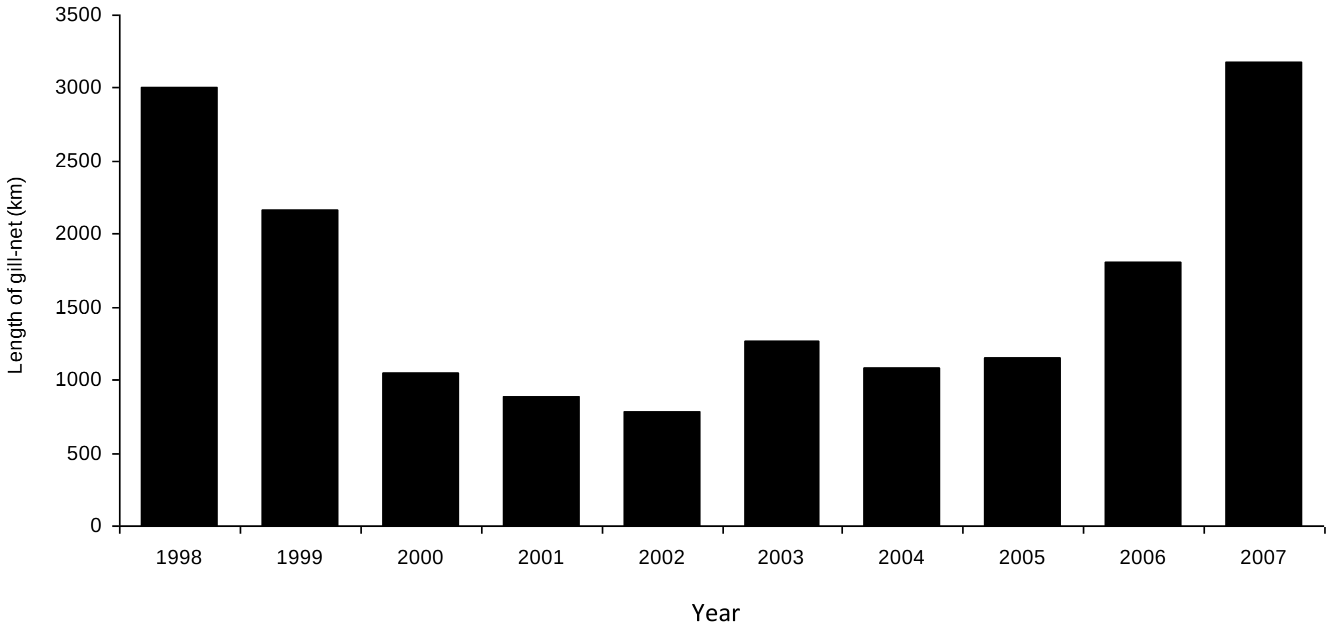


Figure 3 Summary of recorded gillnet fishing effort (km) by year, in the Great Australian Bight region, based on AFMA logbook data.

## 5.6 DISCUSSION

Prior to this study, only 1 other has attempted to assess the effectiveness of spatial management arrangements for protecting Australian sea lions. In WA, bycatch of pups and juveniles in rock lobster pots was mitigated by mandating the use of sea lion exclusion devices (SLEDs) within in-shore waters around breeding colonies (Campbell et al. 2008b). However, our study is the first to specifically assess the effectiveness of an MPA that was designed in part to conserve Australian sea lions. This task proved logistically challenging, because access to the colonies at the base of Bunda Cliffs was dangerous (Dennis & Shaughnessy 1996), thus minimising the number of opportunities to deploy PTTs. Access to vessels also required extensive negotiations (Hamer et al. 2009), because fishers were fearful that interactions with Australian sea lions might jeopardise their future fishing activities, particularly within the GABMP.

During the observer program, 4 Australian sea lion mortalities occurred from 431.4 km of observed set gillnet, equating to 0.0093 mortalities $km^{-1}$. This amounted to between 14 and 33 mortalities during the most recent breeding cycle, between 21 and 42 during the 2 yr study and between 128 and 177 during the 10 yr period since the GABMP was proclaimed. This level of bycatch may have negative implications for the conservation of Australian sea lions and has provided encouragement to explore the benefits of a quantitative risk assessment, with a recent study modelling the impacts of Australian sea lion bycatch in gillnets across all breeding populations in SA (Goldsworthy et al. 2010). However, the results of that study were plagued by a high degree of uncertainty in the data available for Australian sea lions (e.g. life history characteristics and population status) and the fishery (e.g. effort and bycatch incidence), both of which resulted in the need to incorporate assumptions that may have affected the results. Therefore, the quantitative approach to determining the impact of the gillnet fishery on Australian sea lions across SA shelf waters would benefit from the use of more accurate fishing effort and bycatch data from the former and foraging effort data for the latter. Despite these challenges in a relatively data-poor environment, some lessons can be learned from a first principles approach. If it is assumed that there are twice as many sexually mature females in the Bunda Cliffs population as the 161 pups estimated in 1994 (Dennis & Shaughnessy 1996) and the intrinsic growth rate of 4.8% during each breeding cycle reported at Dangerous Reef is realistic (Goldsworthy et al. 2007), then only 13 sexually mature females need be removed in each breeding cycle before population decline becomes imminent. If it is then assumed that half of the 14 to 33 bycatch mortalities (say 7 to 17) estimated in the GAB region during recent breeding cycles are sexually mature females, based on the bycatch data obtained, then the Bunda Cliffs population may be at risk of decline.

The potential risk of population decline highlighted here may be understated, because Australian sea lions that have drowned may drop out of the gillnet below the surface and out of sight of the observer, or those that have become entangled may break free and escape with gillnet material caught around their neck. This situation arises because gillnet meshes are designed to break at 300 to 400 Newtons to facilitate the release of larger, unwanted sharks (Murphy & Richardson 2002). Adult Australian sea lions are of comparable size, and in cases where death occurs, the struggle is likely to have been short due to ensuing panic and associated rapid oxygen depletion. The extent of subsequent entanglement is likely to be minimal, thus increasing the chance that drowned individuals will drop out of the gillnet. It became evident during this study that fishers assumed all entangled animals would be landed on the deck. The fact that this was not the case highlights the need to view observed and reported bycatch as a fraction of the overall numbers drowning in gillnets. It also reinforces the need for fishers and observers to focus their monitoring efforts over the side of the vessel where the gillnet is being hauled rather than on the deck, to increase the chance of detecting a greater proportion of those drowned individuals that at least make it to the surface before dropping out.

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Three of the four recorded bycatch mortalities occurred within the GABMP, because fishing (thus monitoring) took place close to Bunda Cliffs during the open period. Had the GABMP been closed to fishing, the level of bycatch may have been more than halved during recent breeding cycles. This study raises concerns about the effectiveness of the GABMP under its current management arrangements. Rarely have MPAs been proclaimed for a pinniped species (Pires et al. 2008), although their purpose should be to exclude any human activity that deleteriously impacts or threatens the species to be conserved (Hooker & Gerber 2004). It seems incongruous to allow commercial-scale fishing activities, particularly of the nature described in this study, to occur within the GABMP.

The female Australian sea lions tracked from the western and central sites of the Bunda Cliffs, while behaving quite differently, expended a considerable proportion of their foraging effort outside the GABMP (half to three-quarters of their time at sea, two-thirds to most of their foraging trips and 3 to 9 times further than the southern boundary). This is principally because the southern boundary of the GABMP is at most only 21 km south of the coastline. Prior to this study, the only report of maximum distances attained by sexually mature females during foraging trips was about 75 km from Seal Bay (Fowler et al. 2007). Despite this threatening process being permitted within the GABMP for half of the year, the distances travelled by Australian sea lions residing at Bunda Cliffs suggest that they are exposed to the risk of drowning in gillnets year round.

The differences in the at-sea distribution of effort of Australian sea lions from different breeding sites is likely to be linked with the patchy distribution of their benthic prey, which is governed by the heterogeneous structure of the benthos along Australia's southern coastline (Edgar 2008). Unlike pelagic foragers that track highly mobile pelagic prey in a 3-dimensional space (Baylis et al. 2008), Australian sea lions are likely to regularly visit the same structures on the sea floor if the returns are favourable. This study did not allow longer-term, inter-seasonal foraging strategies to be revealed, and thus should be viewed as a snapshot in time. Nonetheless, preliminary investigations of the data obtained for 4 of the animals that were tracked for 7 to 9 mo indicated that little changed in their at-sea behaviour between successive foraging trips, suggesting that prey availability in the benthic environment is relatively stable and predictable (Edgar 2008).

It should be recognised that fishing effort during this study was much lower than between the 1970s and mid-1990s. The late 1990s marked the end of a considerable decline in the activities of the fishery, mainly due to the collapse of the school shark stock (Walker et al. 2005). Therefore, the much higher level of fishing effort in the GAB region during the previous 2 decades would have resulted in much higher estimates of bycatch mortality of Australian sea lions, suggesting that the estimates calculated in this study may understate the longer-term impact of the fishery on Australian sea lions in the GAB region.

### *5.6.1 Summary & recommendations*

This study revealed that Australian sea lions residing at Bunda Cliffs are at risk of drowning in shark gillnets that are set within the GABMP for half of the year and across the GAB region throughout the year. These management arrangements seem contradictory to the purpose and goals of an MPA (Hooker & Gerber 2004). The Australian Government management plan for the GABMP states that it aims to ensure the 'preservation of the area in its natural condition' and to 'protect….habitat for the Australian sea lion', and also acknowledges that the species is listed as vulnerable under the EPBC Act (DEH 2005). The SA management plan uses stronger language, specifically aiming to 'protect and assist in the recovery of….Australian sea lions' (DEHAA 1999).

*

Despite these differences between the 2 management plans, together they continue to permit the use of gillnets across most of the GABMP for half of the year; thus, the likelihood of bycatch mortalities occurring remains high. Highlighting this finding is especially important, because gillnetting was formally identified as a key threat to the conservation of Australian sea lions in 2005 (DEWHA 2008). The findings of this study may be timely, because the current Australian Government plan expires in 2012 although the SA plan can be reviewed at any time in light of new information. As such, GABMP managers are encouraged to amend the language used in the 2 plans to more cohesively protect Australian sea lions residing in the GABMP.

The Australian sea lions tracked in this study revealed that important foraging grounds may also exist outside the GABMP. The only previously published study of the at-sea distribution of effort indicated that sexually mature females foraged up to 75 km from home (Fowler et al. 2007). However, animals tracked from Bunda Cliffs foraged 2.5 times that distance (180 km) and almost 9 times farther south than the southern border of the GABMP. A similar situation existed for Hector's dolphins *Cephalorhynchus hectori hectori* in the Banks Peninsula Marine Mammal Sanctuary (New Zealand), where the distribution of demersal gillnets and of bycatch indicated that the seaward boundary of the sanctuary would need to extend a further 28 km in order to reduce bycatch rates to acceptable levels (Slooten et al. 2006). Using this principle, GABMP mangers should consider extending the southern boundary to more accurately cover the areas utilised by foraging Australian sea lions from Bunda Cliffs, thus more effectively protecting the population from the effects of bycatch mortality.

All of the Australian sea lions reported drowned actually dropped out of the gillnet as they were hauled above the surface, before they could be landed on the deck of the vessel. However, fishers and fishery observers typically concentrate their attention on the deck, in the expectation that all organisms caught in the gillnet will be landed there. This false expectation may explain why the bycatch rates recorded in this study were up to 3 orders of magnitude (i.e.

100s of times) higher than those recorded in AFMA log books. The propensity for drowned animals to drop out of the gillnet at the surface due to the minimal extent of their physical entanglement suggests that an unknown proportion may become caught, drown and then drop out well below the surface, before they can be detected. Others may escape only to die later from injuries caused by gear lodged around their neck (Shaughnessy et al. 2003, Page et al. 2004). This study is the first to reveal that the impact of drowned animals, including those that drop out of the gillnet and those that escape with a gillnet entanglement, may be extensive, both in the GAB region and across the range of the species. This situation should be considered when reviewing the current access arrangements and boundary placement, before the effectiveness of the GABMP in protecting Australian sea lions can be assured.

If fishing is to persist in and adjacent to the GABMP, despite the evidence presented here, then imposing conservative bycatch limits to prevent population decline would be necessary. The concept of potential biological removal (PBR) limits was first introduced in the United States Marine Mammal Protection Act in 1994 and was designed to prevent population decline in situations where human induced mortalities occurred (Wade 1998). The PBR approach was also used to set bycatch limits for the endangered New Zealand sea lion in the Auckland Island squid trawl fishery and resulted in 3 statutory closures and 2 voluntary withdrawals between 1995 and 2000 (Wilkinson et al. 2003). As indicated earlier, the loss of 13 or more individuals as bycatch in each breeding cycle is likely to lead to a population decline at Bunda Cliffs. In order to ensure that the necessarily low bycatch limits are adhered to, high levels of observer coverage may be needed (Lopez et al. 2003, Hamer et al. 2008, Jaaman et al. 2009). In simplistic terms, if the PBR is 7 female Australian sea lions and the level of observer coverage is 50%, then there is a chance that as many as 14 females may be caught before fishing activities cease. Under such a scenario, the Bunda Cliffs population would still be at risk of decline, even though bycatch limits and an observer program were in place. This suggests that either the trigger limits should be reduced in recognition of the uncertainty, or that the level of observer coverage should be increased to reduce the uncertainty.

Based on the findings of this study, the following suggestions for improving the current management arrangements of the GABMP and the conservation outlook for resident Australian sea lions are made:

(1) Year-round exclusion of gillnetting in all zones;

(2) Low bycatch limits;

(3) Extension of the southern boundary further south.

This study has demonstrated that much can be done to improve the effectiveness of the GABMP in protecting Australian sea lions. However, shark gillnetting occurs extensively across the continental shelf adjacent to the SA coastline in both SA and adjacent Australian Government waters, where many other Australian sea lion breeding colonies are located (Hamer et al. 2007, 2010, Goldsworthy et al. 2010). Individuals tend to spend much of their time at or near the benthos (Costa & Gales 2003, Fowler et al. 2006), suggesting that they face some risk of becoming by-caught across much of the area they forage. As such, the conservation of Australian sea lions in the GAB and other regions may benefit from the reduction of bycatch in shark gillnets through carefully placed closures and more stringent monitoring of gillnetting. Population monitoring of key Australian sea lion breeding sites may also be necessary in order to ascertain the long-term benefits of the various initiatives taken by the responsible government agencies. Nonetheless, it will be both necessary and challenging to improve relationships that facilitate appropriate and timely outcomes that benefit Australian sea lion populations in the GABMP and throughout their range, while allowing shark fishing to continue in an economically viable manner.

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## Impact of demersal shark gill-nets on endangered Australian sea lions in South Australia: spatial overlap of fishing and foraging effort and level of by-catch mortality

Prime numbers are what is left when you have taken all the patterns away. I think prime numbers are like life. They are very logical but you could never work out the rules, even if you spent all your time thinking about them.

Mark Haddon.

# Statement of Authorship

Title of publication: The endangered Australian sea lion extensively overlaps with and regularly becomes by-catch in demersal shark gill-nets in South Australian shelf waters

Publication status: Published, December 2012

Publication details: *Biological Conservation* 157, 386-400.

### Author Contributions

By signing the Statement of Authorship, each author certifies that their stated contribution to the publication is accurate and that permission is granted for the publication to be included in the candidate's thesis.

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## 6.1 ABSTRACT

Australian sea lions (*Neophoca cinerea*) have typically small breeding colonies, many of which are genetically distinct populations due to female philopatry (i.e. breeding site fidelity). This situation may increase the species vulnerability to decline when anthropogenic influences increase levels of mortality, even by small amounts. Anecdotal reports from South Australian shelf waters suggest Australian sea lions become by-caught and drown in demersal gill-nets used to catch sharks, or escape with life threatening entanglements. This study explored the potential impact of the operational interaction by estimating the (i) extent of geographic overlap and (ii) level of by-catch. Monitoring of Australian sea lion at-sea movements and of the demersal gill-net fishery confirmed spatial overlap between the two in 68.7% of 4 $km^2$ grid cells across South Australian shelf waters and by-catch of 283 to 333 Australian sea lions each breeding cycle (193 to 227 each year). Recent changes to the management arrangements of demersal gill-netting in South Australian shelf waters are likely to improve the situation for Australian sea lions, although it may be necessary to further refine aspects relating to (i) the effectiveness of untested electronic fishery monitoring methods, (ii) the efficacy of relatively small permanent fishery closures around breeding colonies and (iii) the efficiency in receiving, processing and responding to by-catch reports to ensure limits are not exceeded. Long-term monitoring at representative breeding colonies would be useful for determining if and where research and management should be prioritised. A recent report suggests a similar problem may exist in Western Australia, where approximately 14% of the species resides.

## 6.2 ACKNOWLEDGEMENTS

The authors thank the Fisheries Research and Development Corporation (FRDC), DSEWPaC and the Great Australian Bight Marine Park (GABMP) Steering Committee for providing funds; Kirie, Philios and Lenny Toumazos (The Fish Factory Pty. Ltd.) and Stav Parissos (Theo Parissos and Sons Pty. Ltd.) for access to fishing vessels; skippers Matthew 'Boof' Larsson (FV "Lutarna"), Geoff McDonald (FV "Jean Bryant"), Alan 'Pumper' Austin (FV "Opal Star") and Mark 'Scrubber' Reynolds (FV "Marian H") for support, enthusiasm and insights; Alex Ivey for assistance during the observer program; Clare Fountain, Danielle Gibas, Bec McIntosh, Deb Frazer, Katie Howard, Simon Clark, Alastair Baylis, Kristian Peters, Andy Lowther, Nathan Dowie, Phillip Exton, Guy Abell, Balazs Bajka and Alex Satragno for assistance and good company during field trips to capture Australian sea lions for satellite tracking; Dan Costa (University of California) for Australian sea lion foraging data for Seal Bay; Trent Timmis, Matt Daniel and John Garvey (AFMA), Terry Walker and Matias Braccili (MAFRI) for historical SESSF catch and effort data; Simon Childerhouse (AAD) and at least three anonymous reviewers for assistance with manuscript improvement.

## 6.3 INTRODUCTION

A growing body of research conducted since the early 1990s indicates that Australian sea lions (*Neophoca cinerea*) have low fecundity and their breeding colonies are typically small and unlikely to receive female immigrants due to philopatry (Higgins 1993; Gales et al. 1994; Gales and Costa 1997; DSEWPaC 2012a; Lowther et al. 2012). These characteristics may increase the species vulnerability to decline or extinction when even small increases to the level of mortality occur (Caughley 1994; Goldsworthy et al. 2010; Hamer et al. 2011). Since the late 1960s, a demersal gill-net fishery has operated along the southern Australian coastline (BRS 2004; Walker et al. 2005), overlapping with Australian sea lions most apparently in waters adjacent to South Australia (SA) where the greater proportion of the species resides and forages (Fowler et al. 2007; Hamer et al. 2011; Woodhams et al. 2011; DSEWPaC 2012a). Recent information suggests that Australian sea lions may occasionally become by-caught and drown, or become entangled and eventually succumb from related injuries (Shaughnessy et al. 2003; Page et al. 2004; Goldsworthy et al. 2010; Hamer et al. 2011). The nature, extent and impact of these events remain unclear, thus providing the impetus for this study.

### *6.3.1 Pinniped by-catch: a global perspective*

Since the 1960s, the Southern Ocean has witnessed the recovery and expansion of many pinniped populations, due to the widespread cessation of commercial sealing by the mid 1800s (e.g. Taylor 1982; Roux 1987; Wickens 1995; Kirkwood et al. 2010). Commercial fishing effort has also expanded during the same period, due to technological advances and increased demand for fish (FAO 2009; UN 2009). Consequently, increased overlap between these two marine consumers has resulted in the increased occurrence of direct or ‘operational interactions’ (e.g. Beverton 1985; Woodley and Lavigne 1991; Pemberton et al. 1994; Wickens 1995; Northridge and Hofman 1999;

Hückstädt and Antezana 2003; Shaughnessy et al. 2003; Hamer and Goldsworthy 2006; Hamer et al. 2011). These events occur when marine mammals come into direct or close contact with fishing gear, either intentionally when depredating caught fish, or accidentally when foraging naturally (Northridge and Hofman 1999; Shaughnessy et al. 2003; Read 2005).

Pinnipeds may benefit energetically from depredating fish caught in the fishing gear, although they may also become by-caught and drown when doing so (Northridge and Hofman 1999; Hamer and Goldsworthy 2006), or may escape with life threatening entanglements from which they later succumb (Fowler et al. 1990; Page et al. 2004). The occurrence of these events in demersal gill-nets is widespread and may be the greatest contemporary anthropogenic threat to pinnipeds (Woodley and Lavigne 1991; Wickens 1995; Read et al. 2006; Read 2008). In California, two gill-net fisheries have reported by-catch of four pinniped species (Californian sea lion *Zalophus californianus*, Steller sea lion *Eumetopias jubatus*, northern elephant seal *Mirounga angustirostris* and harbour seal *Phoca vitulina*,; Julian and Beeson 1998). A recent study estimated 98% of all pinniped by-catch in United States of America (USA) commercial fisheries occurs in gill-nets (Read et al. 2006), while another estimated 9% of all California sea lions at one Mexican breeding colony exhibited gill-net entanglements (Aurioles-Gamboa et al. 2003).

Despite widespread occurrence of operational interactions between pinnipeds and fisheries, there have been few attempts to address the problem. Trawl fisheries have received some attention, with New Zealand sea lion (*Phocarctos hookeri*) by-catch mitigated to some extent by applying by-catch limits and temporary closures (Wilkinson et al. 2003), and Australian fur seal (*Arctocephalus pusillus doriferus*) by-catch mitigated by moving away when individuals were observed near the vessel and by including gear modifications (Tilzey et al. 2004; Hamer and Goldsworthy 2006). One lobster trap fishery attempted to mitigate Australian sea lion by-catch by mandating the use of exclusion devices in areas where the species foraged (Campbell et al. 2008). The apparent lack of effort committed to mitigating the impacts of by-catch more widely may be in part due to resistance between the two main stakeholders, with conservationists aiming to protect marine mammals at the expense of the fisheries involved and fisheries aiming to exploit marine resources at the expense of other marine consumers. To date, there are few examples demonstrating a capacity or willingness to adopt a bipartisan approach.

### *6.3.2 Impact of demersal gill-nets on Australian sea lions*

A demersal gill-net fishery has operated along the southern Australian coastline since the late 1960s, targeting benthic dwelling gummy shark (*Mustelus antarcticus*) and school shark (*Galeorhinus galeus*; BRS 2004; Walker et al. 2005). The method used has remained virtually unchanged since its inception, with monofilament polyamide and polypropylene gill-net hung between a weighted foot rope that holds it stationary on the benthos and a floated headline that holds it upright in the water column (Hamer et al. 2011). In waters adjacent to SA, demersal gill-netting is managed by the Australian Fisheries Management Authority (AFMA) in state waters (i.e. from the coastline out to 5.56 km or 3 nm, under a bilateral agreement with the SA Government) and across Australian Government waters (i.e. from 5.56 km out to a maximum permissible depth of 183 m, pursuant to the management arrangements of the fishery), from the SA and Western Australian (WA) boarder, to the Victorian and New South Wales (NSW) border (AFMA 2010; Woodhams et al. 2011). Waters adjacent to SA are particularly important to the fishery, with approximately 40% of effort by km of gill-net hauled occurring there in 2010 (Goldsworthy et al. 2010; Woodhams et al. 2011).

The same waters are also important for the Australian sea lion, with approximately 86% of the species by numbers of individuals and 63% by numbers of colonies residing there (DSEWPaC

2012a). This species is unique when compared with other pinnipeds, firstly by having slow maturation and extended breeding cycles of nominally 17.6 months that reduce overall fecundity by approximately 30% (Higgins 1993; Gales et al. 1994; Gales and Costa 1997). Secondly, colonies are generally small, with 66% of all breeding colonies in SA producing less than 30 pups. Thirdly, females exhibit philopatry, breeding exclusively at their own place of birth and thus unable to facilitate immigration at other sites, which may explain why many breeding colonies or clusters of breeding colonies are genetically distinct (Campbell et al. 2007; Lowther et al. 2012). Collectively, these characteristics may increase the species vulnerability to decline or extinction when even small and unnatural increases in levels of mortality occur (Caughley 1994; Goldsworthy et al. 2010; Hamer et al. 2011; Davidson et al. 2012).

Australia is home to three pinniped species (i.e. Australian fur seal, New Zealand fur seal *Arctocephalus forsteri* and Australian sea lion), all of which have had operational interactions with demersal gill-nets (Shaughnessy et al. 2003). A study during the early 1990s in Tasmania (Australia) found that 15% of entanglements on Australian fur seals involved demersal gill-net material (Pemberton et al. 1992). It is thought that entangled individuals may have been attracted to the benthic fish caught in the gill-nets (Arnould and Kirkwood 2007) that naturally occur in their diet (Deagle et al. 2009). During the early 2000s at Kangaroo Is (SA), 1% of entanglements observed on New Zealand fur seals involved demersal gill-net material (Page et al. 2004). The seemingly low incidence of entanglement may reflect the pelagic foraging habit of the species (Baylis et al. 2008). In contrast, Australian sea lions are known to forage almost exclusively near the sea floor, on benthic prey (Costa and Gales 2003; Fowler et al., 2006). This may explain why the Kangaroo Is study found 55% of entanglements on Australian sea lions involved demersal gill-nets (Page et al. 2004). Given the severity of the wounds resulting from entanglement in demersal gill-nets and the low probability that the material would break away naturally (Peter Shaughnessy, personal communication), an estimated 36 Australian sea lions would die from related injuries each year (modified from Page et al. 2004).

The impact of Australian sea lion by-catch in demersal gill-nets may be evident in population trends at some breeding colonies. Population growth at the Dangerous Reef breeding colony in Spencer Gulf (SA) increased from 0.6% each breeding cycle between 1975 and 2002 to 4.8% each breeding cycle between 2002 and 2007, after a moratorium of shark fishing was issued there in 2001 (SA Government Gazette, 22 March 2001, page 1060-1061; SA Government Gazette, 2 May 2001, page 1703). The Seal Bay population, which is close to an area where demersal gill-netting effort is high, declined at 1.1% each breeding cycle between 1985 and 2003 (Shaughnessy et al. 2006). These examples indicate that Australian sea lion populations are sensitive to the presence of demersal gill-netting activities, and to the additional losses of individuals due to by-catch.

The empirical history of Australian sea lion by-catch in demersal gill-nets has been difficult to determine, because it was not mandatory to record such interactions prior to the enactment of the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) in 2000 (administered by the Australian Government Department of Sustainability, Envrionment, Water, Population and Communities; DSEWPaC). Nonetheless, gill-netting logbook records obtained for the years 2000 to 2007 for the area adjacent to the SA coastline contain records for only 10 drowned seals of unspecified species (Hamer 2007). This low incidence contrasts with two anecdotal reports that suggest high levels of by-catch by individual gill-netters, one of 20 animals being killed each year during the 1990s in southeast SA (Shaughnessy et al. 2003) and the other of about 12 being killed during 2010 (Adelaide Now 2011). A recent independent study, conducted prior to the present study, reported that 7 to 17 female Australian sea lions residing at Bunda Cliffs (western coastline of SA) were by-caught and drowned in demersal gill-nets set in and adjacent to the Great Australian Bight Marine Park (GABMP) each breeding cycle, with only an estimated 13 needed to suffer this fate before population decline would be imminent (Hamer et al. 2011). Therefore, by-catch of Australian sea lions in demersal gill-nets is likely to occur regularly and be geographically widespread.

### *6.3.3 Protection measures for Australian sea lions*

Partial legal protection for Australian sea lions first occurred in WA in 1892 and in SA in 1919, although unregulated killing for skins and for fish bait continued into the 1970s (Thiele 1979; Ling 1999; Shaughnessy 1999). Currently, Australian sea lions are listed as 'Vulnerable' under the SA *National Parks and Wildlife Act 1972* (SADEH 2011) and as 'Specially Protected' under the WA *Wildlife Conservation Act 1950* (WA Government Gazette, 17 February 2012, page 746), forming the basis for protection inside state waters, where all breeding colonies are located. In response to growing concerns about the impact of entanglement and by-catch in demersal gill-nets in SA and the adjacent Australian waters, the Australian Government listed the species as 'Vulnerable' pursuant to the EPBC Act in 2005 and the International Union for the Conservation of Nature (IUCN) included the species as 'Endangered' on the Red List of Threatened Species in 2008 (Goldsworthy and Gales 2008; Hamer et al. 2011). Pursuant to the EPBC Act, DSEWPaC must facilitate a recovery plan for a Vulnerable species, identifying priorities and actions for mitigating the impacts of the threatening processes, or activities (DSEWPaC 2012b).

All major fisheries in Australia are required, pursuant to the EPBC Act, to obtain a conditional Wildlife Trade Operation (WTO) permit to harvest a native fish species and thus are also required to minimise their impacts on the marine environment when doing so. Renewal of the WTO permit is dependent on the permitted fishery lodging a mandated environmental assessment (EA) and on addressed recommendations arising during the previous permit period. The demersal gill-net fishery in southern Australia is part of the broader Southern and Eastern

Scalefish and Shark Fishery (SESSF), which conducted its first EA in 2003. The EA highlighted the potential for operational interactions between Australian sea lions and demersal gill-netting, prompting DSEWPaC to recommend that AFMA (i) “establish a robust reporting system” and (ii) “if necessary, trial and implement appropriate mitigation measures such as spatial closures” (DEH 2003). Based on emerging information, in 2010 DSEWPaC released new conditions on the then current WTO permit. Condition 6b required AFMA to “implement long-term management measures, including formal closures and other actions, that lead to a significant reduction of the impact of fishing activity on Australian sea lions. These measures [should] be clearly directed towards enabling recovery of the species, including all subpopulations” (Australian Commonwealth Government Gazette, 19 February 2010, page 1-4).

Mitigating the impact of demersal gill-nets on Australian sea lions could be effected by reducing the degree of spatial overlap between the two, namely by implementing spatial closures or marine protected areas (MPAs), thus excluding demersal gill-netting in areas of critical habitat for Australian sea lions. A number of MPAs in SA and adjacent Australian waters offer some protection to Australian sea lions. The GABMP, proclaimed in 1998, is the largest MPA along the SA coastline and extends up to 21 km seaward from Bunda Cliffs (Hamer et al. 2011). The remaining MPAs, such as the one adjacent to Seal Bay and proclaimed in 2009, extend only 5.6 km out to sea (DENR 2009). These coastal MPAs are unlikely to provide adequate protection to resident Australian sea lions, because recent tracking studies indicate that females forage much further offshore at distances of 77 to 193 km from their natal colonies (Fowler et al. 2007; Hamer et al. 2011).

### *6.3.4 Aims of this study*

The available evidence suggests that Australian sea lions regularly become by-caught and drown in demersal gill-nets in areas where the two physically overlap. This phenomenon may threaten the conservation of Australian sea lion populations. As such, the use of satellite telemetry to determine the movements of pinnipeds (e.g. Baylis et al. 2008; Hamer et al. 2011) and of observers to monitor the movements and activities of commercial fishing vessels (e.g. Bastardie et al. 2010) could assist in quantifying levels of overlap and by-catch and facilitate quantitative popualiton viabiilty analysis (PVA), which can ultimately be used by managers seeking to conserve Australian sea lions. Therefore, this study aimed to estimate (i) the spatial distribution and geographic overlap between Australian sea lions and the demersal gill-net fishery from satellite telemetry and fishery logbooks and (ii) the level of Australian sea lion by-catch by monitoring demersal gill-net fishery activities.

## 6.4 METHODS

### *6.4.1 Australian sea lion foraging effort*

The at-sea movements of adult female Australian sea lions were determined at 16 of 49 known breeding colonies across waters adjacent to the SA coastline (Fig. 1). Desirable attributes of the selected breeding colonies (SBCs) included relative ease of access and broad geographic spread across the species range. Sexually mature females were the focus of the foraging study, because they produce offspring and exhibit philopatry and thus are directly linked to the sustainability or vulnerability of populations.

To facilitate deployment of satellite linked platform transmission terminals (PTTs; Mark I & II, Sirtrack, Havelock North, New Zealand), females were captured and restrained using a purpose built cone-shaped net, then sedated with isoflorane (Isoflo™, Veterinary Companies of Australia, Artarmon, New South Wales, Australia) delivered with oxygen via a vaporiser (Cyprane Tec III, Advanced Anaesthetic Specialists, Melbourne, Victoria, Australia). A PTT was placed along the mid-dorsal line 10 cm posterior of the fore-flipper pits and attached to guard hairs using rapid curing two part epoxy adhesive. Devcon 5 Minute® (ITW Devcon, Massachusetts, USA) was used at Seal Bay and Araldite® 2017 (Huntsman Advanced Materials, Basel, Switzerland) was used at all other SBCs.

Location data were obtained from Services Argos Inc (Toulouse, France). Class A, B and Z locations were removed from the dataset prior to analysis due to their inaccuracy (following: Sterling and Ream 2004; Costa et al. 2010). In addition, data linked to locations within 1 km of the natal colony were excluded to account for error margins and the potential inclusion of resting animals, thus confining analyses to at-sea positions only. The remaining data were redistributed into 1 $km^2$ grid cells across SA shelf waters using specifically written script (*timeTrack* and *trip*) for use in the software package R (Version 2.3.0, R Foundation for Statistical Computing, Vienna, Austria). For this study, 'shelf waters' adjacent to the SA coastline are defined as those waters under SA Government jurisdiction (i.e. from the coastline out to 5.6 km offshore) and those under Australian Government jurisdiction (from 5.6 km seaward to the maximum depth contour of 183 m), collectively covering approximately 178 000 $km^2$.

The resulting values were then used to calculate the overall proportion of foraging effort in each 1 $km^2$ grid cell for each individual, then standardised by determining the fractional contribution of (i) each tracked individual to the overall tracking time calculated for all tracked individuals from an SBC, (ii) each SBC to the overall tracking time calculated for all SBCs and (iii) each SBC to overall pup abundance in SA shelf waters. Analysis of these values was facilitated using MapInfo Professional® and Vertical Mapper® (Versions 9.0 and 2.5, respectively, MapInfo Corporation, New York, USA) and the interpolation function (i.e. triangulation with smoothing) was used to identify areas of high, intermediate and low foraging effort.

For greater clarity, the results were presented in maps across four broad regions, referred to in this study as Bunda, Nuyts, Eyre and Kangaroo, based on geographically distinguishable regions containing clusters of breeding colonies (Fig. 1). For each SBC, summaries were presented for mean foraging distance (based on the straight line distance between each recorded location received for each tracked individual and the location of its associated SBC), mean maximum foraging distance (MMFD; based on the straight line distance between the furthest recorded location of each foraging trip received for each tracked individual and the location of the associated SBC) and mean direction of travel (based on the direction of the location of each tracked individual recorded closest to midday each day). In addition, the percentage of time spent in SA waters, in adjacent Australian waters and in each AFMA Marine Fishery Area (MFA;

effectively degree by degree cells) was calculated. Effort has traditionally been recorded by MFA, thus calculations were additionally made at this coarser scale to provide fishery related context.

### *6.4.2 Demersal gill-net fishing effort*

Shark gill-net fishing effort data were obtained from AFMA logbooks between 1 January 2000 and 31 December 2008. Prior to 1 July 2007, shark gill-net fishers recorded the location of fishing events by MFA. After that time, it became mandatory to more accurately record location by degrees and minutes (for latitude and longitude), although this has mostly occurred since 1 January 2006. As such, the location-based fishing effort for the three year period between 1 January 2006 and 31 December 2008 were used to determine areas of high, intermediate and low gill-netting effort across SA shelf waters, using MapInfo Professional® and Vertical Mapper®. This was done at an aggregated 4 $km^2$ grid cell scale, to account for the 1.8 km error in all directions caused by the prevalent absence of the geographic *second* value in the spatial component of the fishery logbook data. Additionally, to maintain an agreed level of commercial confidentiality in locations where the number of fishing events or fishing effort inside a given MFA was low (which occurred several times in the Bunda region), the location of fishing events was centralised to the nearest 6 minute central node (i.e. one tenth of a degree). While the metric for recording Australian sea lion foraging effort was time, most of the fishery effort data recorded between 1 January 2006 and 31 December 2008 was recorded by event. Although the 'soak time' (i.e. the amount of time that the gill-net is in the water during each fishing event) could have been standardised for each event and thus made proportional to the number of fishing events, this was deemed unwise (thus not attempted), due to the wide variation in fishing strategies known to be in use in the fishery.

### *6.4.3 Overlap & by-catch estimates*

The level of geographic overlap between tracked Australian sea lions and demersal gill-netting activities was calculated by multiplying the proportion of time spent by Australian sea lions by the proportion of km of gill-net set in the aggregated 4 $km^2$ grid cells. The percentage of cells utilised by both Australian sea lions and shark gill-netters was also calculated to provide a simplistic indication of spatial overlap, without an index of relative effort. Based on available data (Costa and Gales 2003; Fowler et al. 2006; Goldsworthy et al. 2010), it was assumed that all of the adult females tracked had the capacity to dive to and forage on the sea floor across all SA shelf waters, thus had the potential to encounter demersal gill-nets in all the areas they visited.

Independent scientific observers accompanied demersal gill-net vessels in SA shelf waters between 1 January 2006 and 31 December 2007 to monitor fishing activities. The observers employed the technique described in Hamer et al. (2011) of vigilantly observing from outboard of the gunwale, as the net was hauled through the upper several metres of the water column to the surface and then onto the net roller. Specifically, records included the time and location the gear was hauled at the end of a fishing event, the number Australian sea lions by-caught and drowned in the gill-net and the number that dropped out before being hauled aboard. Where possible, the age and gender of the by-caught individual was determined by inspecting the genitalia, size and pelage colouration. The colour of the gill-net in which by-caught individuals were caught was also recorded.

The observed by-catch mortality rate for Australian sea lions was calculated by dividing the number of individuals observed drowned by the number of km of gill-net that were observed hauled. The estimated number of Australian sea lions drowned in demersal gill-nets was then calculated by multiplying the observed by-catch mortality rate by overall gill-net effort across SA shelf waters for the two calendar years, then halved to obtain the number for one calendar year. The number was then multiplied by 1.47 to obtain an estimate for a nominal 17.6 month breeding cycle. The standard error of the annual and cyclic by-catch estimates (SEE) was then calculated to estimate the level of precision, or variance, resulting from the expected low level of observer effort. This was calculated as:

$$SEE = \sqrt{1 - \left(\frac{observer\ effort}{fishing\ effort}\right)} \cdot mortality\ estimate$$

where the square root of one minus the sample fraction, based on kilometres of gill-net set, was multiplied by the mortality estimate. To achieve 95% confidence that the upper and lower level of the two by-catch estimates were accurate, the SEE was multiplied by two to reflect two standard deviations (Cochrane 1977).

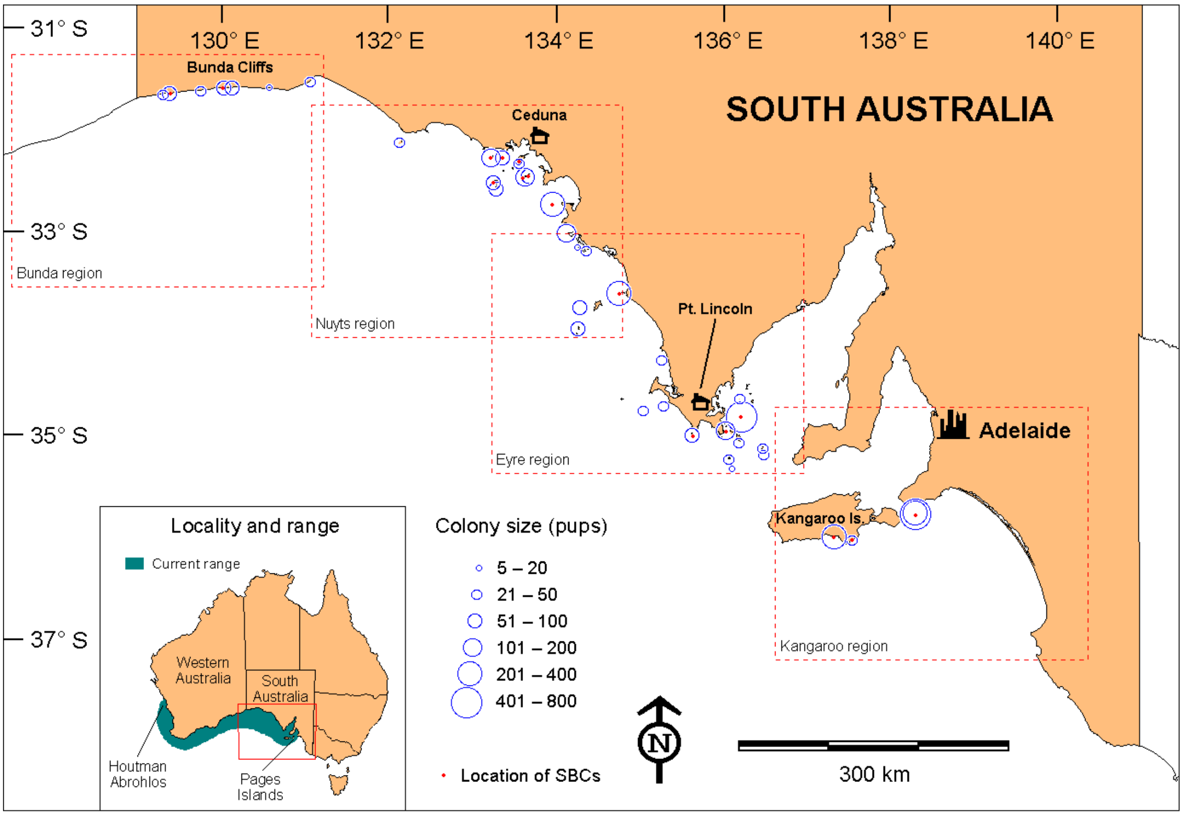


Figure 1 Location map, showing the (i) overall range of Australian sea lions (insert, green area), (ii) location and numbers of pups at known breeding sites in South Australian shelf waters (blue circles), (iii) the 16 selected breeding colonies (SBCs) that were the focus of this study (red dots) and (iv) four regions referred to in this study (red dashed boxes).

## 6.5 RESULTS

### *6.5.1 Australian sea lion foraging effort*

A total of 115 adult female Australian sea lions was tracked from the 16 SBCs (1.8% of the total number at SA breeding colonies), yielding 4590 days (individual: $\tilde{x}$ = 39.9; SD = 54.0) of tracking information. Nine animals from two SBCs in the Bunda region were tracked for 1264 days (individual: $\tilde{x}$ = 140.4; SD = 119.6), 41 from seven SBCs in the Nuyts region were tracked for 1500 days (individual: $\tilde{x}$ = 36.6; SD = 37.5), 25 from four SBCs in the Eyre region were tracked for 714 days (individual: $\tilde{x}$ = 28.6; SD = 20.5) and 40 from three SBCs in the Kangaroo region were tracked for 1112 days (individual: $\tilde{x}$ = 27.8; SD = 36.3).

The MMFDs achieved by tracked animals varied widely, from 28 ± 18 km at Lewis Is to 189 ± 25 km at Bunda-8 (Table 1). The direction of travel generally ranged between south-westerly and south-easterly, with the exception of South Pages Is where some individuals travelled in a north-westerly direction (Figures 2a, 3a, 4a, 5a). The percentage of time spent in each MFA visited by tracked animals also varied widely, with MFA-108 being the most utilised at 86.1% after standardisation of tracking effort (Table 2). In summary, tracked animals foraged in 20 of the 29 MFAs and across 27.9% of the 4 $km^2$ grid cells in SA shelf waters (26.9% of SA waters and 28.3% of Commonwealth waters).

### *6.5.2 Demersal gill-net fishing effort*

Over the nine years between 2000 and 2008, 153 800 km of shark gill-net was set, with effort in all 29 MFAs across SA shelf waters, amounting to an annual average of 17 089 km (SD = 2238). Higher levels of effort occurred along the west coast of the Eyre Peninsula in MFAs 108 and 115

(12.2%) and to the south-east of Kangaroo Is in MFAs 150 and 151 (30.4%; Fig. 6). The least effort occurred within Spencer Gulf (in MFAs 122, 129 and 132) and in Gulf St Vincent (MFA 136; 0.7% collectively), where shark gill-netting has been banned since 2001. During the three years between 2006 and 2008, when the more accurate location data were collected, a total of 52 064 km of demersal gill-net was deployed, amounting to an annual average of 17 354 km (SD = 852). Based on the more recent and more accurate data, higher levels of effort occurred south of Bunda Cliffs, along the west coast of the Eyre Peninsula and to the south-east of Kangaroo Is (Fig. 2b, 3b, 4b, 5b).

### *6.5.3 Overlap & by-catch estimates*

There was considerable overlap of Australian sea lion foraging effort and shark gill-netting effort in all four regions across SA shelf waters. In the Bunda region, the greatest level of overlap occurred offshore around 130° E, where MFAs 101, 102, 104 and 105 intersect (Fig. 2c). In the Nuyts region, overlap was more widespread and occurred throughout most of the islands of the Nuyts Archipelago, across MFAs 107, 108 and 114 (Fig. 3c). In the Eyre region, overlap was patchier and occurred in MFAs 115, 126 and 138 (Fig. 4c). In the Kangaroo region, extensive overlap occurred throughout waters to the south and south-east of Kangaroo Is, in MFAs 149 and 150 (Fig. 5c). Overall, Australian sea lion foraging effort and shark gill-net fishing effort overlapped in 68.7% of the 4 $km^2$ grid cells across SA shelf waters.

Observer data were collected over 146 days at sea across SA shelf waters between 1 January 2006 and 31 December 2007. A total of 994.4 km of gill-net was observed hauled (from 234 fishing events), which equates to 2.9% of the combined length of gill-nets set across SA shelf waters during the two year monitoring period. Twelve Australian sea lions were observed by-caught and drowned during that period, equating to an overall by-catch rate of 0.01207 individuals per km of gill-net (or 0.05128 per fishing event). Therefore, based on the calculated annual fishing effort of 17 355 ± 852 km of deployed gear recorded in fishery logbooks and taking the SEE into account, an estimated 193 to 227 Australian sea lions drowned annually in demersal gill-nets across SA shelf waters during the study period, or 283 to 333 each 17.6 month breeding cycle. The gender of 10 of the by-caught individuals was determined, with 90% being females. As such, an estimated 174 to 204 females were drowned annually, or 255 to 299 each breeding cycle. The mean distance of the 12 observed by-catch mortalities to the nearest breeding colony was 12.6 km (SD = 13.8), with nine occurring within 10 km. Four occurred in the Bunda region, with three close to breeding colonies at Bunda Cliffs and one 46.3 km to the south (Fig. 2c). Eight occurred in the Nuyts and Eyre regions, with six close to breeding colonies and two 20.4 km and 32.6 km away (Fig. 3c, 4c).

Ten of the 12 (83%) by-caught individuals dropped out of the gill-net before they were hauled aboard the vessel, either as they ascended above the waterline, or as they made contact with the net roller. The two individuals brought aboard the vessel were small juveniles, suggesting the weight of the 10 larger individuals increased the probability of structural failure of the meshes in the gill-net. Seven of the by-caught individuals occurred in green gill-net, four occurred in pink gill-net and one occurred in white gill-net, being 58.3, 33.3 and 8.4%, respectively. Unfortunately, the sample size of the data collected by observers did not allow robust analyses, nor were fishers required by AFMA to record gill-net colour in their logbooks, thus it was not possible to determine the relationship between gear colour and the likelihood of by-catch.

**Table 1**

The location, mean maximum foraging distance (MMFD) and mean direction travelled by 115 adult female Australian sea lions tracked from 16 selected breeding colonies (SBCs) across South Australian (SA) shelf waters, by region and by colony.

| | Regions | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Bunda | | Nuyts | | | | | | | Eyre | | | | Kangaroo | | |
| | Bunda-8 | Bunda-5 | Purdie | West | Lounds | Breakwater | East Franklin[a] | South Franklin[b] | Olive | West Waldegrave | Liguanea | Lewis | Dangerous | Seal Bay | Seal Slide | South Page |
| *Colony location* | | | | | | | | | | | | | | | | |
| Latitude (dec. °) | 31.640°S | 31.585°S | 32.270°S | 32.511°S | 32.273°S | 32.322°S | 32.449°S | 32.462°S | 32.719°S | 33.596°S | 34.998°S | 34.957°S | 34.817°S | 35.997°S | 36.026°S | 35.777°S |
| Longitude (dec. °) | 129.381°E | 130.031°E | 133.228°E | 133.251°E | 133.366°E | 133.561°E | 133.669°E | 133.639°E | 133.970°E | 134.762°E | 135.620°E | 136.032°E | 136.217°E | 137.327°E | 137.536°E | 138.292°E |
| *Foraging summary* | | | | | | | | | | | | | | | | |
| MMFD (km) | 189 | 67 | 64 | 39 | 33 | 35 | 32 | 108 | 48 | 49 | 28 | 30 | 28 | 67 | 59 | 98 |
| MMFD SD | 25 | 11 | 27 | 27 | 29 | 11 | 14 | 38 | 27 | 23 | 18 | 3 | 13 | 15 | 42 | 30 |
| Mean dir. (° true) | 206 | 185 | 247 | 162 | 99 | 199 | 138 | 197 | 128 | 186 | 189 | 237 | 179 | 171 | 98 | 248 |
| Mean dir. SD | 22 | 18 | 26 | 66 | 53 | 110 | 88 | 27 | 79 | 90 | 65 | 38 | 38 | 25 | 28 | 59 |

[a] Officially named Lilliput Is in 2008.
[b] Officially named Blefuscu Is in 2008.

**Table 2**
The percentage of time spent by 115 adult female Australian sea lions tracked from 16 selected breeding colonies (SBCs) across South Australian (SA) shelf waters, by region, by colony and by Australian Fishery Management Authority (AFMA) Marine Fishery Area (MFA).

| Management zone | Colony | | |
|---|---|---|---|
| | Bunda-8 | Bunda-5 | Combined |
| *a. Bunda region* | | | |
| MFA-101 | 49.3 | 32.8 | 44.4 |
| MFA-102 | | 52.1 | 15.4 |
| MFA-104 | 50.7 | 7.4 | 37.9 |
| MFA-105 | | 7.7 | 2.3 |
| South Australian waters | 7.5 | 21.0 | 11.5 |
| Adjacent Commonwealth waters | 92.5 | 79.0 | 88.5 |

| Management zone | Colony | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Purdie | West | Lounds | Breakwater | E Franklin | S Franklin | Olive | Combined |
| *b. Nuyts region* | | | | | | | | |
| MFA-107 | 52.4 | 14.1 | 0.1 | | | | | 5.2 |
| MFA-108 | 47.6 | 85.9 | 99.9 | 100.0 | 100.0 | 69.5 | 93.8 | 86.1 |
| MFA-114 | | | | | | 30.4 | 6.2 | 8.6 |
| MFA-115 | | | | | | 0.1 | | 0.1 |
| South Australian waters | 45.3 | 68.9 | 99.9 | 100.0 | 99.6 | 48.5 | 81.5 | 76.9 |
| Adjacent Commonwealth waters | 54.7 | 31.1 | 0.1 | | 0.4 | 51.5 | 18.5 | 23.1 |

| Management zone | Colony | | | | |
|---|---|---|---|---|---|
| | W Waldegrave. | Liguanea | Lewis | Dangerous | Combined |
| *c. Eyre region* | | | | | |
| MFA-115 | 94.2 | | | | 30.4 |
| MFA-126 | 5.3 | | | | 1.7 |
| MFA-128 | 0.5 | 43.3 | | | 9.7 |
| MFA-129 | | | 100.0 | 89.9 | 41.4 |
| MFA-138 | | 56.7 | | | 12.5 |
| MFA-139 | | | | 10.1 | 4.3 |
| South Australian waters | 69.7 | 44.1 | 100.0 | 100.0 | 77.9 |
| Adjacent Commonwealth waters | 30.3 | 55.9 | | | 22.1 |

| Management zone | Colony | | | |
|---|---|---|---|---|
| | Seal Bay | Seal Slide | S Page | Combined |
| *d. Kangaroo region* | | | | |
| MFA-140 | | | 47.8 | 44.5 |
| MFA-144 | | 0.2 | 23.9 | 20.3 |
| MFA-148 | 22.8 | | | 1.5 |
| MFA-149 | 77.2 | 91.9 | 0.3 | 13.3 |
| MFA-150 | | 7.8 | 27.9 | 24.4 |
| MFA-151 | | | 0.1 | 0.1 |
| South Australian waters | 86.2 | 26.8 | 68.4 | 66.0 |
| Adjacent Commonwealth waters | 13.8 | 73.2 | 31.6 | 34.0 |

a.

b.

c.

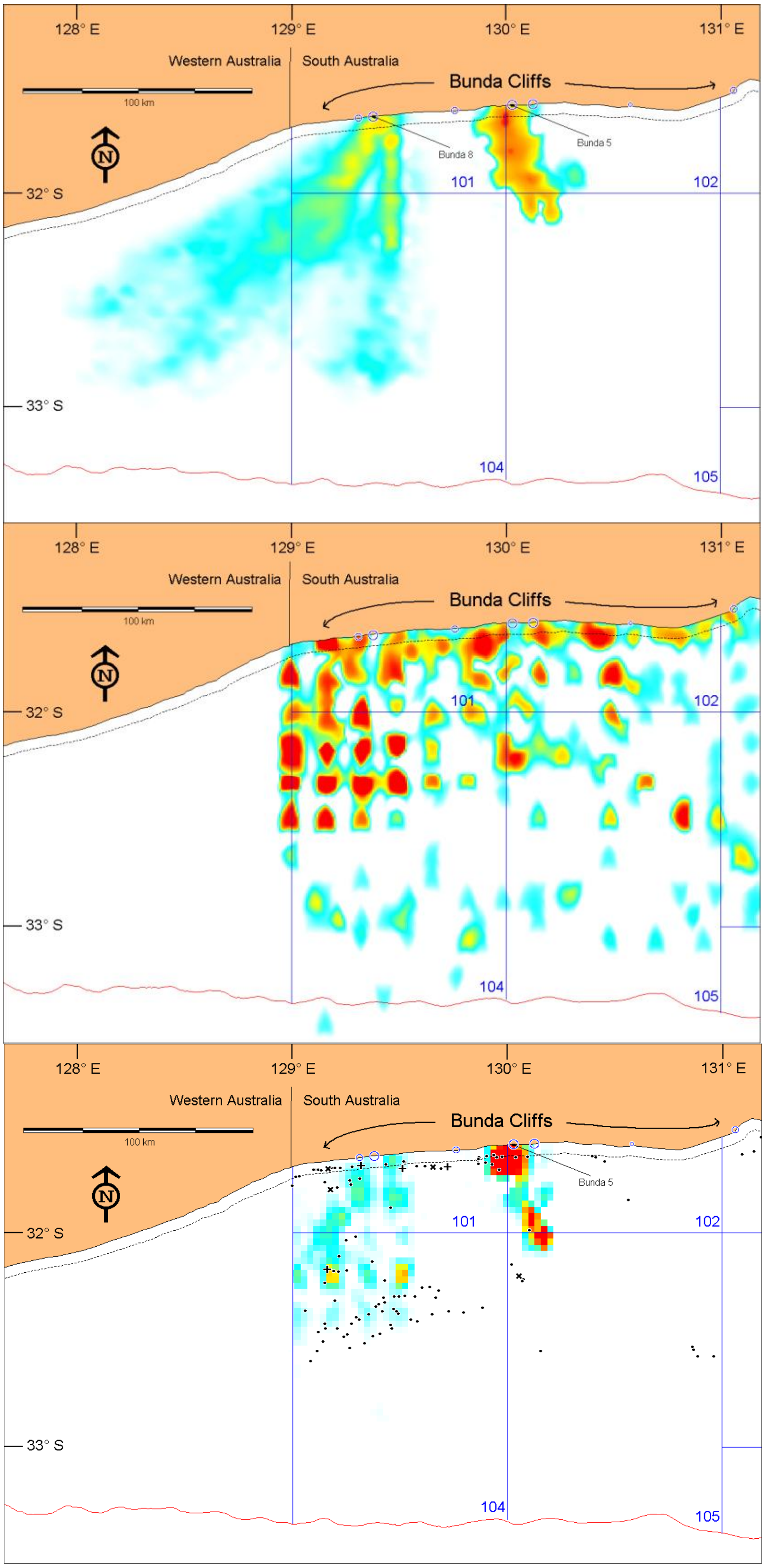


Figure 2 *Effort distribution map in the 'Bunda region'*. Showing (a) at-sea movement of nine sexually mature Australian sea lion females tracked from two selected breeding colonies (SBCs) in 2006-07, (b) demersal shark gill-net fishing activity in 2006-08 and (c) overlap between the two (effort/overlap: red = high effort, orange = medium, blue = low). Location of observed fishing activity (.), presence of an Australian sea lion during hauling (x) and by-catch mortality (+) are also marked.

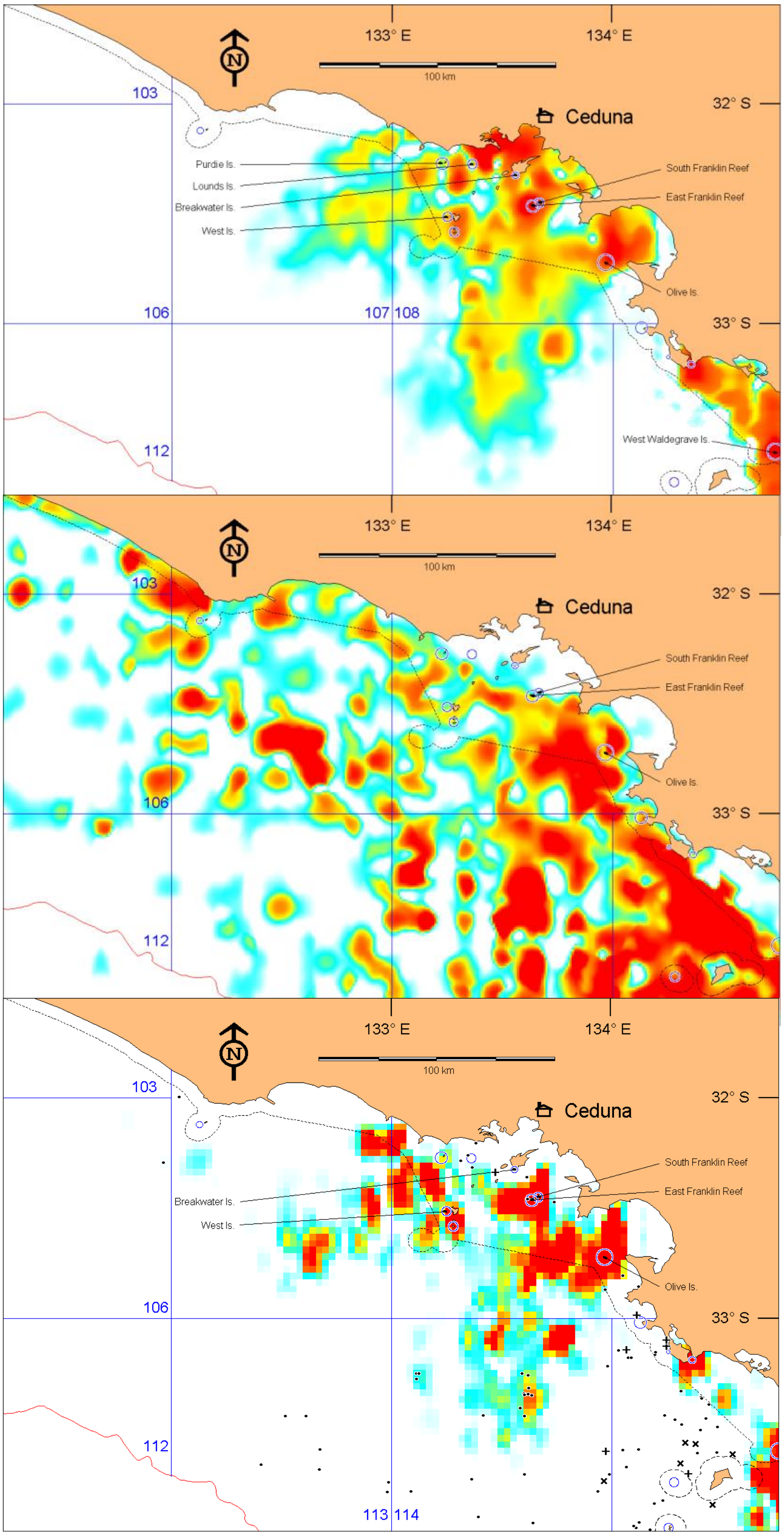


Figure 3 *Effort distribution map in the 'Nuyts region'*. Showing (a) at-sea movement of 41 sexually mature Australian sea lion females tracked from seven selected breeding colonies (SBCs) in 2006-07, (b) demersal shark gill-net fishing activity in 2006-08 and (c) overlap between the two (effort/overlap: red = high effort, orange = medium, blue = low). Location of observed fishing activity (.), presence of an Australian sea lion during hauling (x) and by-catch mortality (+) are also marked.

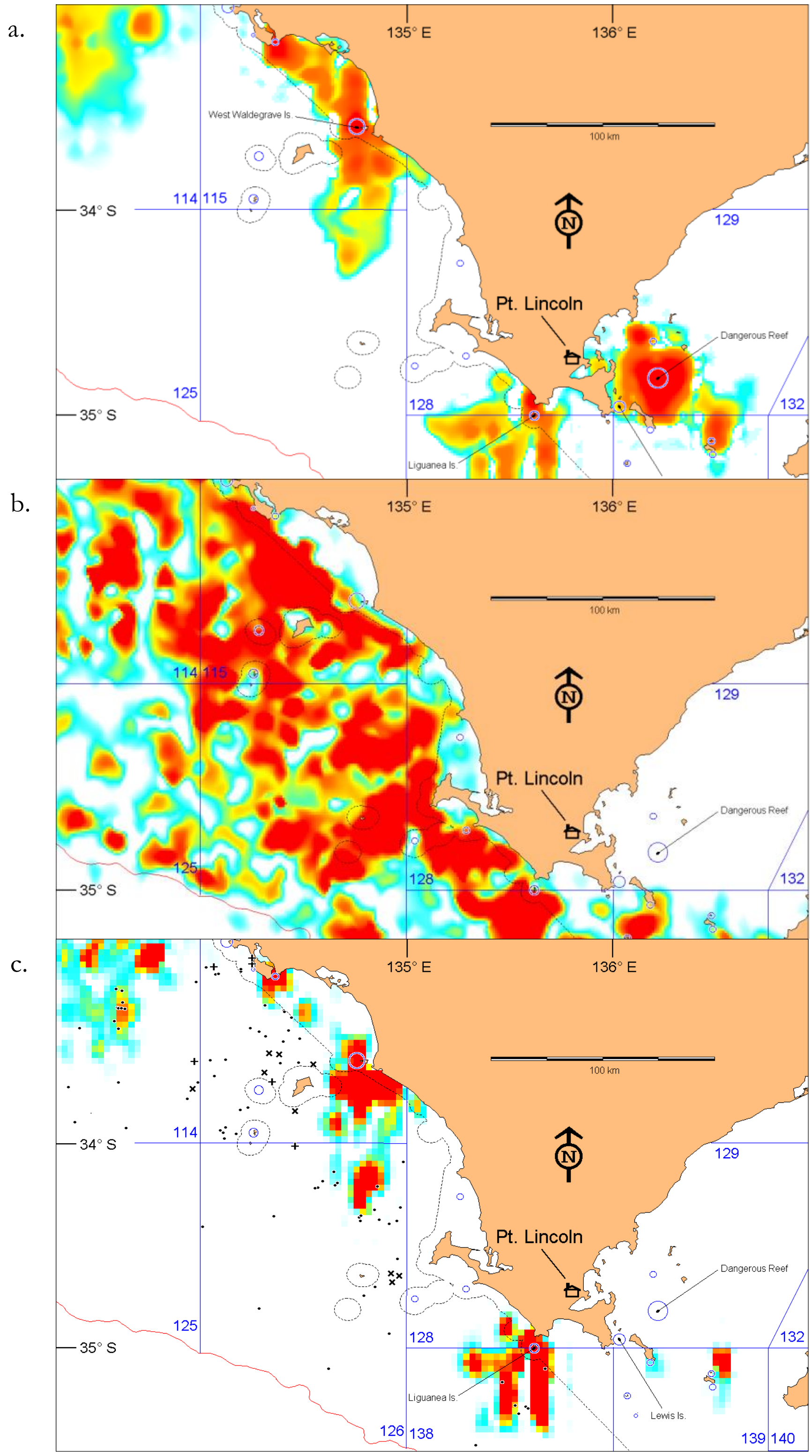


Figure 4 *Effort distribution map in the 'Eyre region'*. Showing (a) at-sea movement of 25 sexually mature Australian sea lion females tracked from four selected breeding colonies (SBCs) in 2005-08, (b) demersal shark gill-net fishing activity in 2006-08 and (c) overlap between the two (effort/overlap: red = high effort, orange = medium, blue = low). Location of observed fishing activity (.), presence of an Australian sea lion during hauling (x) and by-catch mortality (+) are also marked.

a. b. c.

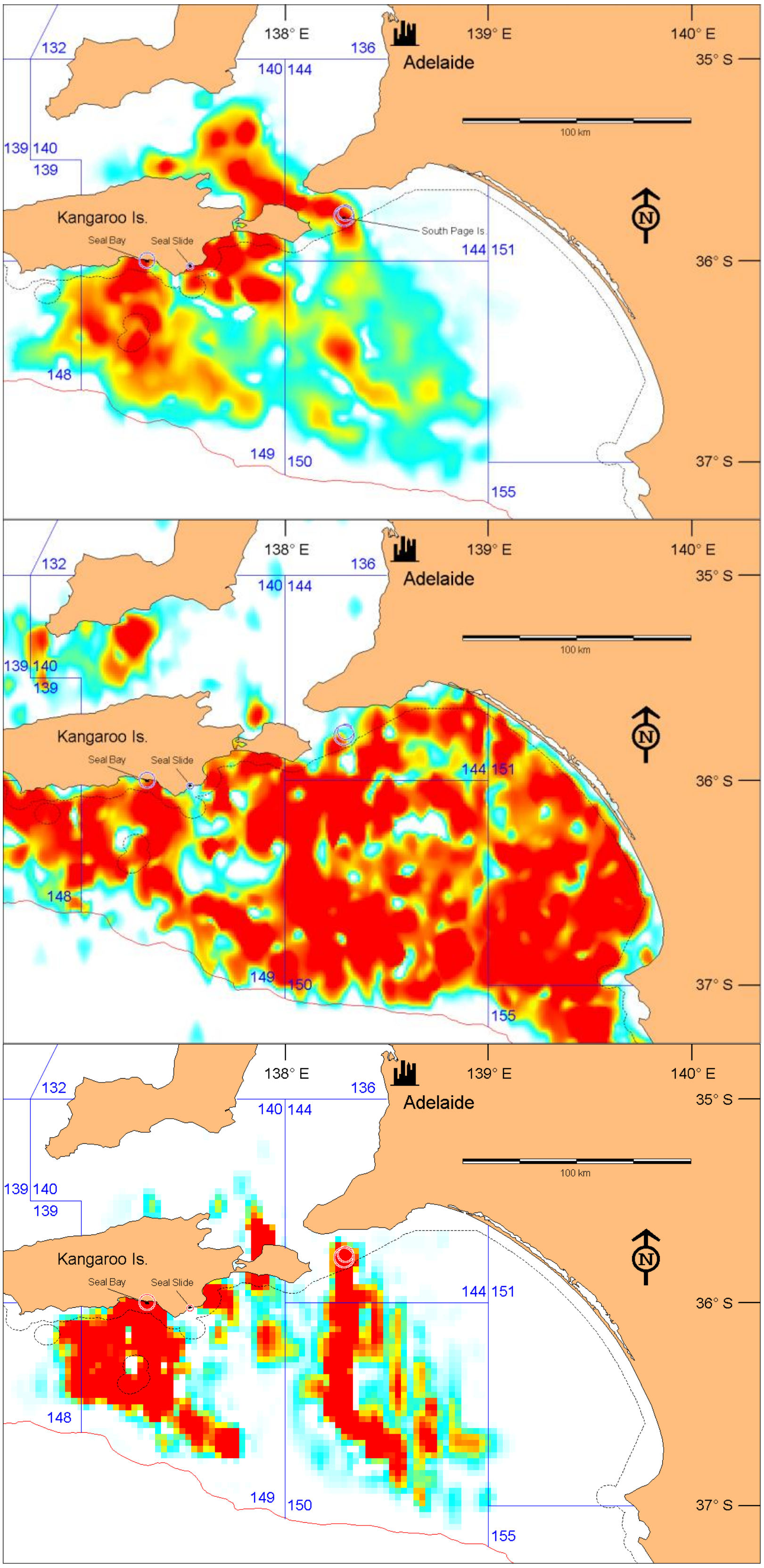


Figure 5 *Effort distribution map in the 'Kangaroo region'*. Showing (a) at-sea movement of 40 sexually mature Australian sea lion females tracked from three selected breeding colonies (SBCs) in 2002-06, (b) demersal shark gill-net fishing activity in 2006-08 and (c) overlap between the two (effort/overlap: red = high effort, orange = medium, blue = low).

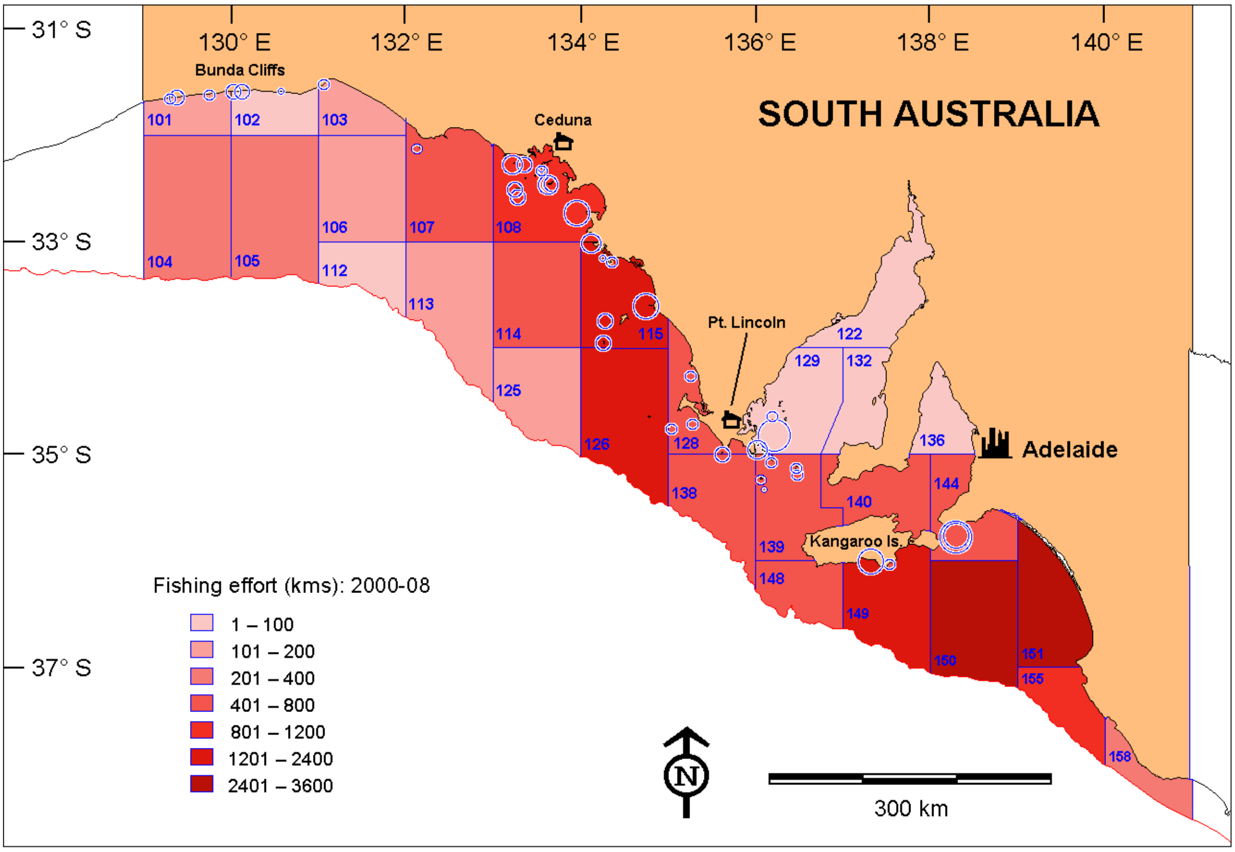


Figure 6 *Distribution of demersal shark gill-net fishing effort by MFA*. Across South Australian shelf waters in 2000-08, shown as blue numbered degree by degree grid cells. Location of known breeding sites (blue circles) shown for reference (blue circles; see our Fig 1 for size reference).

## 6.6 DISCUSSION

### *6.6.1 Widespread at sea distribution of Australian sea lions*

The satellite tracking of adult female Australian sea lions reported in this study represents the most comprehensive investigation of at-sea behaviour of the species to date. Specifically, this study involved 1.8% of adult females from 16 of the 48 breeding locations in SA waters, across approximately 1 100 km of coastline from near the boarder with WA at Bunda Cliffs to the eastern end of the species range at South Page Is. At sea effort was geographically widespread, covering 27.9% of the approximately 178 000 km$^2$ area of SA shelf waters, as defined in this study. The extensive utilisation of SA shelf waters is attributable in part to the broad distribution of the 16 SBCs and to the generally diverse although individually specialised foraging strategies exhibited at a colony scale (Hamer et al. 2011; Lowther et al. 2011). Individuals from some colonies foraged inshore to MMFDs of only 28 km from their natal colony, while others foraged offshore to MMFDs of 189 km, or six to seven times the distance. These greater distances confirm that Australian sea lions can travel much further than previous reports indicate (Fowler et al. 2007). The fact that foraging tracks did not extend further than the relatively shallow SA shelf waters suggests that Australian sea lion suggesting foraging effort is probably limited by sea floor depth (Goldsworthy et al. 2010) and by the suite of prey species found there. The variation between colonies likely demonstrates cultural differences that are developed and sustained over long periods, with individuals learning where and how to forage from other individuals in older cohorts (Lowther et al. 2011). Given that 32 other breeding colonies are located in SA waters, where many more females reside, it is likely they collectively utilise most of the SA shelf waters in coastal and offshore regions.

### *6.6.2 Overlap of Australian sea lions & demersal gill-nets: potential impact of by-catch*

Given the extensive geographic distribution of demersal gill-netting and of Australian sea lions across SA shelf waters, the extensive overlap recorded (i.e. 68.7% in 4 $km^2$ grid cells) seems intuitive, although this should be viewed as a minimum estimate due to the small proportion of females tracked. Predictive analyses using generic models for Australian sea lion foraging behaviour are also likely to confirm extensive overlap (Goldsworthy et al. 2010), although this study adequately demonstrates that geographic overlap is prevalent in SA shelf waters wherever gill-netting takes place, albeit to varying degrees. Although this study is limited to the 16 SBCs, where in some cases only a small percentage of animals were tracked, it should be noted that predictive analyses using generic foraging models to determine geographic overlap across SA shelf waters (Goldsworthy et al. 2010) may be inaccurate, because of the inter and intra colony variance in the spatial distribution of foraging effort. This poses significant problems for effective conservation management on a finer scale, where it is important to know which areas are utilised the most.

Other evidence external to this study suggests vertical overlap between demersal gill-nets and Australian sea lions may also be extensive. School and gummy sharks occur close to the sea floor in temperate shelf habitats where demersal gill-nets are set (Walker et al. 2005; Hamer et al. 2011) and Australian sea lions also concentrate their foraging efforts close to the sea floor, seemingly from as soon as they leave the colony or haulout site (Costa and Gales 2003; Fowler et al. 2007; Baylis et al. 2009; Goldsworthy et al. 2010). As such, Australian sea lions and demersal gill-nets are likely to overlap vertically in the water column, as well as geographically. This situation further highlights the increased risk of Australian sea lions becoming by-caught, because they either fail to see the fishing gear while foraging naturally, or become entangled and drown while attempting to depredate fish caught in the gill-net.

*

Based on the most recent overall estimate of the number of Australian sea lions residing in SA waters being 14 780 (Shaughnessy et al. 2011) and the estimated range of overall by-catch being 283 to 333 individuals each breeding cycle, the SA component of the species could be losing 1.91 to 2.25% of its individuals each breeding cycle. This unnatural and additional source of mortality may serve to increase the risk of local decline or even extinction if left unabated (Goldsworthy et al. 2010). This is especially relevant when considering that Australian sea lions exhibit a comparatively low level of fecundity and have many small colonies that are probably genetically distinct populations (Higgins 1993; Gales et al. 1994; Campbell et al. 2007; Lowther et al. 2012). The Bunda Cliff population in the GABMP was reported to lose an estimated 7 to 17 female Australian sea lions as by-catch in gill-nets each breeding cycle, with the loss of only 13 required to exceed the intrinsic (i.e. naturally possible) rate of population growth, suggesting that local population decline could be occurring (Hamer et al. 2011). Nonetheless, small numbers have persisted there for at least 20 years, since at least the early 1990s (Dennis and Shaughnessy 1996; Hamer et al. 2009). It is also possible that higher levels of by-catch may have occurred there and in other locations during the 1980s, at a time when the level of fishing effort was greater (Woodhams et al. 2011). Assuming population declines did occur earlier on due to the additional by-catch related mortalities, the level of by-catch would also have declined. Given that the likelihood of by-catch is proportional to foraging density, levels of by-catch would eventually become sufficiently low to no longer be the principal cause of decline. Alternatively though, any increase in population size would also have been prevented under a reversal of the same process, assuming all other factors remain unchanged. In a situation where gill-netting and Australian sea lion by-catch is commonplace, these two opposing forces may act to stabilise the size of the population at artificially lower numbers. This situation may explain the lack of extinction events at the many small breeding colonies in SA waters, even though they are disproportionately exposed to the effects of stochastic events, such as disease (e.g. New Zealand sea lion epizootics; Robertson and Chilvers 2011) and human development (e.g.

recreational boating: Gales et al. 1994; aquaculture: Goldsworthy et al. 2009). As such, small and seemingly stable colonies are more likely to rapidly and unexpectedly go extinct if circumstances change even slightly for the worse. Such events are without recourse because females exhibit philopatry, thus preventing the immigration of females from other breeding sites to facilitate recolonization.

Independent observer programs are widely accepted as the most practical method for monitoring by-catch, from which rates and estimates can be calculated (e.g. Read 2005; Gilman 2011). In this study, monitoring occurred in approximately half of the 29 MFAs available for demersal gill-netting across SA shelf waters and by-catch occurred in approximately half of those where monitoring occurred, in both coastal and offshore environments. More by-catch data is needed confirm if the probability of an Australian sea lions becoming by-caught is greater in coastal waters (i.e. close to breeding colonies) than in offshore waters, although it is likely given that the density of at-sea traffic of adult females is likely to be greatest near breeding colonies, because these central place foragers need to regularly return to feed nutritionally dependent pups (e.g. Chilvers et al. 2011). However, spatial overlap in SA shelf waters between Australian sea lions and demersal gill-netting was found to be extensive in both coastal and offshore waters, suggesting that the few individuals by-caught in offshore waters may not be an anomaly. It should also be noted that the extent of the problem may be underestimated, because only observed by-catch was included in the calculations. The novel monitoring technique used by observers during this study confirmed that the majority of drowned animals dropped out of the gear as they breached the surface before they could be hauled aboard the vessel. These events would normally go undetected and thus unreported by crew, thus not recorded by conventional fishery observers who tend to focus their attention on the deck where caught fish are processed. It seems likely that the weight of the drowned animal and a sudden increase in gravity as it emerges from the water may cause the gill-net meshes to break, thus allowing the by-caught individual to fall back into the water where it remains unnoticed. This phenomenon also raises questions about the proportion of drowned individuals that may drop out of the net as it is hauled up off the sea floor, which cannot be detected using the technique developed for this study. This problem affects many fisheries that have problems with marine mammal by-catch (Warden and Murray 2011). However, at present there are no reliable or practical methods of observing what occurs in a demersal gill-net while it is actively fishing on the sea floor. Therefore, it must be assumed that individuals observed by-caught and drowned are only a portion of the actual level of by-catch.

Reports of Australian sea lions observed entangled at breeding sites confirm some by-caught individuals manage to break free of actively fishing demersal gill-nets with an entanglement, without drowning immediately (Shaughnessy 1999; Shaughnessy et al. 2003; Page et al. 2004). Other studies have suggested that individuals observed with entanglements at breeding sites are a very small portion of the true number entangled (Fowler 1987; Fowler et al. 1990). This may be due to entangled individuals seeking to avoid interactions with other individuals at breeding colonies that may result in further injury and to the need to forage for longer periods to compensate for the energetic inefficiencies caused by the entanglement. Given that demersal gill-nets are made of thin but durable monofilament polyamide or polypropylene (Walker et al. 2005) and that entanglement related injuries are typically extensive (Raum-Suryan et al. 2009), it is probable that many individuals that initially escape with an entanglement will eventually die a slow and painful death. This eventuality also has implications for nutritionally dependent pups, which are likely to starve and die if their mother is lost as by-catch or from associated entanglement injuries.

### *6.6.3 Current management approaches to mitigating Australian sea lion by-catch*

In response to the findings documented in this study and to preliminary PVAs conducted in another recent study (Goldsworthy et al. 2010), AFMA implemented the Australian sea lion Management Strategy (MS), which focuses on three elements for mitigating the impact of the demersal gill-net fishery on Australian sea lions in SA shelf waters (AFMA 2010). Firstly, vessel-based monitoring was increased from negligible levels to 100%, to more accurately determine the level of Australian sea lion by-catch (AFMA 2011). However, the extensive use of electronic camera systems as part of the monitoring effort may be premature, because their accuracy has not yet been adequately compared with human observers in this fishery. This process is unlikely to be a swift, because a sizeable dataset would be required to facilitate pair wise statistical comparisons. Nonetheless, recent effort in a Canadian hook-and-line fishery suggests random audits using cameras are sufficiently accurate to verify logbook data when used under stringent conditions (Stanley et al. 2011), although their application to rare by-catch remains to be tested. Secondly, AFMA implemented year-round area closures that extend 7.3 to 20.7 km in all directions around the 48 breeding sites in SA (AFMA 2011). However, the tracking results of this study demonstrate they are unlikely to reduce geographic overlap sufficiently to prevent by-catch, thus some small populations may remain at risk of decline, or even extinction. Thirdly, if the by-catch limits of 1 to 5 allocated in each of the seven large AFMA designated zones across SA shelf waters are reached, then much larger closures are implemented out to the boundary of the fishery in the associated zone (i.e. the 183 m depth line) for 18 months, which approximates one breeding cycle (AFMA 2011, 2012a). However, recent information suggest by-catch limits are being exceeded (AFMA 2012b), presumably due to delays in receiving, processing and responding to by-catch reports. Despite the need for continued improvement in AFMAs MS, the changes made to date are substantial and are likely to improve the conservation situation for Australian sea lions.

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Comparable approaches toward managing the impact of major commercial fisheries on a pinniped species are scarce in the published literature. One example from New Zealand is relevant, where by-catch limits of New Zealand sea lions and associated area closures have formed part of the management plan for the Auckland Is arrow squid (*Nototodarus* spp.) trawl fishery since the early 1990s (Wilkinson et al. 2003). However, a recent proposal to remove by-catch limits was justified by citing negligible by-catch in recent years (MAF 2011). This decision seems incongruous when the reduction in by-catch would have been linked (at least in part) to the by-catch limits, the use of exclusion devices and the associated area closures. This example highlights the need to properly interpret the reduction in recorded by-catch, by first determining if it is caused by the mitigation strategies originally implemented, or by an overall decline in the population.

### *6.6.4 Summary & suggestions for improved conservation management*

This study demonstrates that Australian sea lions forage across a large proportion of SA shelf waters and have extensive geographic overlap with shark gill-netting activities. Given that both target prey species at or near the benthos, it is not surprising that Australian sea lions regularly become by-caught and drown in demersal gill-nets. The level of by-catch reported in this study is likely to represent a fraction of the overall occurrences, with many drowned animals dropping out of the net and going unobserved and many escaping with entanglements only to die later from related injuries. It is possible, with time, that small colonies or populations that are already affected (and thus reduced in size) by by-catch related losses may be exposed to a stochastic event that could cause further declines and possibly even extinction.

The key aspects of Condition 6b of the WTO for the demersal shark gill-net fishery ostensibly called for its impact on Australian sea lions to be addressed and mitigated. While AFMA has taken considerable steps to address these recommendations through the implementation of the MS, further improvements may include:

1. Determining the accuracy of electronic camera monitoring systems in detecting Australian sea lions, before permitting their widespread use on demersal gill-net vessels fishing in SA shelf waters,
2. Expanding the permanent closures currently implemented around 48 breeding sites to further reduce the degree of overlap between Australian sea lions and demersal gill-nets, thus further reducing the likelihood of by-catch,
3. Ensuring that reports of by-catch are more swiftly received, processed and responded to in order to minimise the chance that specified by-catch limits are not exceeded.

Although outside the scope of the MS, long-term monitoring of Australian sea lion population levels and trends at key breeding sites would also be useful in tracking the overall status of the species and its populations into the future. Making decisions about which sites to monitor should be based on region and size representation, although the adverse impact of monitoring activities should be a consideration when developing survey strategies and when conducting activities in small breeding colonies. Nonetheless, population status and trajectories should not be used as a tool to assess the effectiveness of management changes to the demersal gill-net fishery, because many other external and possibly unquantifiable factors are likely to influence recovery rates. A good example of this is the continuing non-recovery of formerly abundant populations of dolphins in the eastern tropical Pacific, where there have been substantial (orders of magnitude) reductions in the by-catch mortality of intentionally targeted dolphin pods by purse-seiners in search of associating tunas (Wade et al. 2007). Therefore, long-term monitoring would be used most effectively to inform conservation and fishery managers about where and when to prioritise Australian sea lion conservation efforts at a colony, region, or species scale, rather than a tool for assessing the effectiveness of fishery management arrangements.

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Future research opportunities that may enhance the management of Australian sea lions include more thorough investigation of gene flow (especially of males) between colonies and regions to identify possible management units (Lowther et al. 2012) and of PVA using density dependent factors and different spatial management scenarios (Goldsworthy et al. 2010). A recent report confirms that low levels of Australian sea lions by-caught in demersal gill-nets in WA may have reduced the population size of many colonies there to low levels and may put them at further risk of decline and extinction (Campbell 2011). Therefore, the conservation of the Australian sea lion would benefit from determining the impact of demersal gill-net activities on populations in WA.

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# General discussion

## 7.1 Overview of key findings

Presented in this thesis are five case studies exploring the problem of operational interactions between marine mammals and fisheries, manifest as by-catch and depredation. Chapter 2 (published as: Hamer et al. 2012) reviewed odontocete (i.e. toothed whales, such as the false killer whale *Pseudorca crassidens*, pilot whale *Globicephala* spp. and melon headed whale *Peponocephala electra*) by-catch and depredation in demersal and pelagic longline fisheries. After defining the relevant terms (e.g. depredation and by-catch) and outlining the background and context, the available relevant literature was reviewed. The phenomenon was found to be geographically widespread, occurring in all major oceans and involving at least 20 odontocete species. The literature indicated that the impacts on the odontocetes and fisheries involved both positive (i.e. odontocetes expended less energy depredating and fishers used odontocetes to find fish) and negative (i.e. odontocetes became by-caught and were injured or drowned, and catch returns for fishers were diminished by depredation). The most commonly pursued avenues for mitigation of by-catch and depredation were acoustic deterrence strategies. These also received the most developmental attention, although with mixed success. Although in their infancy, there have been some promising developments in the use of physical deterrence as a practical alternative, or as a tool to be used in conjunction with acoustic deterrence.

Chapter 3 (published as: Hamer and Childerhouse 2012) specifically characterised by-catch and depredation of odontocetes in Australian and Fijian pelagic longline fisheries, then developed and trialled two devices designed to deter depredating odontocetes and thus mitigate these two problems. Given the equivocal results associated with acoustic technology in recent years, it was decided to explore the use physical deterrence based on advice from fishers that odontocetes avoided tangles in the fishing gear when depredating. Preliminary results of two developmental physical deterrence devices simulating gear tangles were promising. However, only a few depredation and by-catch events occurring during the sea trials; frustratingly, the results could not be used to quantitatively assess the performance of the two devices in mitigating by-catch and depredation. Of the few depredation and by-catch events that did occur, all were on control fishing gear and not on the fishing gear to which the two types of deterrence devices were attached. Additionally, neither of the two types of devices had an adverse impact on target fish catch rate, size or survival, nor on the speed of the fishing operation. The positive results associated with these operational elements are important, because they are likely to encourage ongoing commitment from fisheries toward continued development of the two devices and ultimately for striving to achieving the objective of mitigating odontocete by-catch and depredation.

Chapter 4 (published as: Hamer et al. 2008) characterised common dolphin (*Delphinus delphis*) encirclement and mortality in a South Australian (SA) purse-seine fishery for sardines (*Sardinops sagax*), and assessed the performance of changes to fisher behaviour through a Code of Practice (CoP). Prior to implementing the CoP, an independent observer program revealed high numbers of dolphin encirclements, with some dolphins thought to have died due to physical trauma and drowning in net folds beneath the vessel, while others were thought to have died due to the stress of being encircled. The CoP stipulated that fishers must delay the setting the gear when dolphins were observed and swiftly release dolphins that became encircled, which a subsequent observer program revealed had substantially reduced the encirclement and mortality rates. In response to these findings, the CoP has become mandatory in the fishery and independent observer records and fisher logbook records are compared to ascertain the reliability of the latter (Hamer et al. 2009a).

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Chapters 5 (published as: Hamer et al. 2011) and 6 (published as: Hamer et al. in press) characterised the extent and nature of the operational interaction between Australian sea lions (*Neophoca cinerea*) and a demersal gill-net fishery operating in SA. Specifically, independent observer programs were conducted to ascertain Australian sea lion by-catch rates and satellite tracking data was obtained from a number of females and overlayed with fishery logbook data. Chapter 5 found that the management arrangements of the Great Australian Bight Marine Park (GABMP) were likely to be inadequate for protecting resident Australian sea lions at Bunda Cliffs, because fishing had been allowed within most of the GABMP for half of the year and because some of the individuals tracked spent a large proportion of their time foraging in areas beyond its borders. The Australian Government Environment Department (DSEWPaC) has subsequently proposed additional closures in concert with the GABMP under the National Marine Parks Network that will restrict gill-netting across a larger area (e.g. DSEWPaC 2012). Additionally, the Australian Fisheries Management Authority (AFMA; the regulatory authority managing the shark gill-net fishery) implemented a year-round 4 nautical mile exclusion zone along Bunda Cliffs, which in some areas almost quadruped the level of protection provided previously by the GABMP (AFMA 2010). Chapter 6 found that Australian sea lions utilise a large proportion of SA shelf waters, resulting in extensive overlap with demersal gill-net fishing activities and a level of by-catch that may be unsustainable for some breeding colonies. In response, a series of fishery management arrangements were implemented that now prohibit gill-netting close to all known breeding colonies in SA and that impose further substantial closures if prescribed by-catch limits are reached (AFMA 2010, 2011, 2012a, 2012b). Although considered to have improved the situation for Australian sea lions, ongoing monitoring of population trends in the long term may be the only method of confirmation.

## 7.2 Potential challenges to the successful development of mitigation strategies

The following four sections explore some of the more obvious challenges associated with developing strategies to mitigate marine mammal by-catch and depredation in fisheries. Specifically, they are (i) the challenge of obtaining statistical accuracy and precision when recording rare events, (ii) the importance of reliability in fishery logbook data, (iii) the intelligence of marine mammals and the difficulty in developing effective mitigation strategies and (iv) the typically limited protection offered by marine parks to this highly mobile and wide-ranging group of species. Although there are likely to be other challenges, these are thought to me of most interest and relevance when embarking on similar activities elsewhere.

### *7.2.1 Rare events: effect of imprecision on management decisions*

Although the rarity of by-catch would seem positive, it may still increase a population's susceptibility to decline and extinction. A large population in equilibrium, where birth rates match death rates, will initially decline because of the additional deaths associated with fishery by-catch. This may have little impact on the conservation of a large population in the short term, because there are still plenty of breeding animals in the population and thus plenty of births. Additionally, the initial decline may be 'equalised' or 'neutralised' over time, because there are fewer animals in competition for the same important and limited resources, thus increasing foraging, breeding and recruitment success. However, unabated by-catch may be problematic in the long term if the level of by-catch continues to exceed the innate capacity of the population to produce enough young and for those young to survive to adulthood and breed. Populations that are naturally smaller, due to smaller quantities of important resources, are more susceptible to decline and extinction, because they have less time and capacity to adjust to the increased death rate caused by fishery by-catch. This risk has been highlighted for the small

Hectors dolphin (*Cephalorhynchus hectori*) population in New Zealand that are by-caught in gill-nets (e.g. Slooten 2006) and more recently for Australian sea lions in chapters 5 and 6.

Despite the potential impact of low levels of by-catch (and depredation), it is often difficult to obtain statistical precision. In most instances there are considerable funding and time constraints when attempting to characterise the problem or assess solutions, because of the protracted timeframes required to document the typically rare events. As such, obtaining a statistically meaningful result may not be possible in many cases. Instead, it is often only possible to collect 'snapshots' of data that from a small part of the geographical area of interest, from small proportions of the animals in a population or across a region, or fishing vessels in a fleet. Under such circumstances, by-catch may be missed altogether, or documented so few times as to render almost impossible the chance of characterising the problem, let alone arrive at a definitive figure or level that is representative for the entire population or region. In some cases, this may result in levels of by-catch that range from having negligible to catastrophic impacts on a population (e.g. Hamer et al. 2009c).

The rarity of an event highlighting the potential imprecision in the data collected, rather than expressing the gamut of possible outcomes. In this thesis, for of the five chapters highlight the difficulties in estimating the impact of by-catch and the efficacy of mitigation strategies:

1. *Chapter 3*. Odontocete by-catch was absent throughout the controlled experimentation of the two developmental mitigation devices in a tuna pelagic longline fishery, which prevented their efficacy from being tested and the concept from being proved, despite the monitoring of 40,514 hooks, plus the expenditure of approximately US$350,000.
2. *Chapter 4*. Despite the reduction in the dolphin by-catch rate after the introduction of a code of practice (CoP) to the SA sardine purse seine fishery, it was not possible to determine if it was due to the CoP directly, or an alleged localised depletion in sardines causing dolphins to forage elsewhere. This was despite the monitoring of 138 fishing

events over 178 nights, across two 7-month periods. Additionally, the by-catch rates were derived from a small proportion of the overall fishing effort, suggesting that extrapolated estimates may not have been representative for other areas and other times.

3. *Chapters 5 and 6*. The apparent degree of overlap between Australian sea lions and a demersal shark gill-net fishery may have been distorted by the considerable variability in foraging behaviour within and between the breeding sites from which animals were tracked. This was despite the monitoring of 234 nights of gill-net fishing and tracking of 115 adult female Australian sea lions over 4590 days, at a cost of about US$500,000. Again, the small amount of fishing effort that was observed suggests that extrapolated by-catch estimates may not be representative.

It is often difficult for fishery managers to justify precautionary decisions designed to mitigate negligible or imprecise levels of by-catch. The desire, oftentimes driven by vocal fishers resiting changes that may result in greater restrictions to fishing activities, is to collect more data over longer time periods, across larger areas, and from more animals and vessels. As this thesis demonstrates, this approach does not guarantee definitive results. However, growing pressure from increasingly aware community and conservation groups have encouraged the use of the precautionary principle, suggesting that management decisions should be made based on the degree of precision or imprecision. This is especially true when estimating levels of marine mammal by-catch (and the conservation impact they may have), thus facilitating timely management approaches in the absence of more precise figures (Brook 2002). Interestingly, although not surprisingly, the imprecision in the by-catch estimates calculated in chapters 5 and 6 were a source of criticism among fishers when faced with the ensuing management recommendations. Delaying subsequent management decisions, while almost invariably suiting the economic needs of the fishers and fisheries involved, may cause irreversible damage to the conservation of the marine mammal populations and species involved. Therefore, fishery managers are encouraged to collectively weigh up the advantages and disadvantages of making decisions that are reliant on ‘best estimates’ in the short term (e.g. Brook et al. 2002), compared with the continued use of practices that may threaten the conservation of a marine mammal population while waiting for a ‘more representative’ suite of data to emerge (e.g. Ellner et al. 2002).

Two additional and unintended problems may emerge if continuing to collect data in the search for definitive estimates. Firstly, small marine mammal populations may be exposed to undue disturbance when collecting more data on population size, life history characteristics, or foraging behaviour. This in itself may become a conservation threat, especially if foraging or breeding processes are interrupted (e.g. Casper 2009; McMahon et al. 2012; Walker et al. 2012). Secondly, where a population is known to be small and declining, the extra time required to collect more data may delay much needed precautionary protection measures, which in turn may increase the risk of that population going extinct (e.g. Slooten et al. 2000, 2006; Grech and Marsh 2008; Davidson et al. 2012). Again, all stakeholders responsible for the welfare of or the threat to marine mammal populations are encouraged to take precautionary action based on limited and often imprecise data and estimates, rather than maintaining the status quo while attempting to increase sampling effort, in the hope that subsequent analyses will become more statistically robust and provide more precise estimates.

Some researchers have used quantitative population viability analyses (PVAs) to deal with the imprecision caused by the rarity of by-catch and predict its impact at a population scale. This and similar modelling approaches are attractive to researchers, because they can be rapidly developed using informed assumptions and fitted with the available data (e.g. Brook et al. 2000). However, there is a risk that available data sets may be too small to guarantee accuracy, regardless of the level of precision, and that the assumptions used may be prejudiced by the user’s perception of the situation (e.g. Taylor 1995; Ellner et al. 2002). As such, while PVAs offer a sound method for predicting the impact of a fishery on a marine mammal population, their implementation may sometimes be problematic. For example, a spatially oriented PVA was developed to predict the response of Australian sea lions to seemingly low levels of by-catch in demersal shark gill-nets (see chapters 5 and 6 for background), with the results predicting for many smaller populations that only one or two additional fishery induced mortalities need occur to cause population decline and eventual extinction (Goldsworthy et al. 2010). However, the models were based on very limited data life history data from on of 49 known breeding populations (from McIntosh 2007), did not account for density dependent responses (e.g. Herrando-Perez 2012) and assumed that each population was clearly defined and closed (Lowther et al. 2012). Additionally, the spatial component of the models was based on highly variable foraging data (see chapter 6), which is unlikely to be representative of individual populations. The implementation of such models may have profoundly erroneous effects on the response of each population to increased death rates. Such approaches may result in inadequate conservation measures that do not sufficiently protect the marine mammal population involved, or alternatively in extreme conservation measures that have unnecessary economic consequences for the fishery involved. Whatever the case, it is important to note that, as in chapter 6, small populations are at risk extinction due to the amplified effects of stochastic events, which may be more rapid than PVAs may predict. Therefore, it is again important to acknowledge that precautionary measures should be taken to protect small populations, regardless of whether or not sound PVAs have been conducted.

The approach taken in this thesis has been, where appropriate, to use predictive analyses to gain a broader understanding of the impact of seemingly rare by-catch events on marine mammal populations, or on rare but reputedly increasing depredation on a fishery. Nonetheless, care has been taken to 'tailor' recommendations that acknowledge the level of imprecision in the data, thus supporting defendable and precautionary actions and outcomes.

### 7.2.2 *Fishery logbook data: reliability and independence*

Fishing activities are often at odds with conservation goals for marine mammals and other elements of the marine ecosystem (Lewison et al. 2004). Fishers that have operational interactions with marine mammals are often reluctant to report incidences of by-catch because they are fearful of the consequences, such as negative public response and increased restrictions on their activities and decreased revenue as a result (e.g. Roman et al. 2011). The levels of odontocete and pinniped by-catch reported by independent observers for the studies presented in chapters 3 to 6, across three different fisheries, were sufficiently high to warrant initiating mitigation efforts, using either voluntary or mandatory strategies. However, fishery logbook records prior to each study suggested that by-catch was either negligible or absent, suggesting underreporting may have been occurring. Additionally, after the conclusion of the study in chapter 4 where observers monitored dolphin by-catch before and after the introduction of a CoP, subsequent records indicated continued underreporting of by-catch in fishery logbooks despite the extensive exposure of the fishery to their regulatory responsibilities and obligations (Hamer et al. 2009a).

The presence of observers on fishing vessels provides an opportunity to convey information about the importance of reliable logbook recording for effective management of the fishery and its impact on elements of the broader ecosystem. Unfortunately though (as revealed in the studies presented this thesis), the presence of observers does not always guarantee fisher compliance, with fishers often only reporting incidences of by-catch when observers are on board (e.g. 'the observer effect'; Wahlen and Smith 1985; Burns and Kerr, 2008). Even fisheries that claim to be stringently managed under the auspices of international fishery conservation instruments are known to suffer such problems, such as the Commission for the Conservation of Southern Bluefin

Tuna (CCSBT; e.g. Polacheck 2012). As a result, reliance on fishery logbook data for reporting by-catch of threatened species has been largely abolished in US fisheries (Andrew J. Read, personal communication).

There is a growing perception, mostly in developed countries, that marine mammals should be conserved for their aesthetic and ecosystem value. However, failure to accurately report when and where marine mammal by-catch and depredation occurs hinders effective management in two fundamental ways, because conservation managers may fail to determine when protection measures for marine mammal populations are warranted, because the reported levels of by-catch are lower than is actually the case (e.g. Roman et al. 2011). It should be acknowledged though that some element of by-catch remains unobservable, as discussed in chapters 5 and 6, although its characterisation can assist in developing correction factors when calculating by-catch estimates. Alternatively, observed by-catch can be viewed as minima, due to an unknown proportion of by-catch gong unobserved and possibly escaping with life threatening injuries or entanglements (e.g. Warden and Murray 2011).

Based on the studies presented in this thesis and elsewhere, it should be assumed that fishery lobook data is an unreliable source of information for calculating marine mammal by-catch rates and estimates. Nonetheless, some elements of logbook data, such as the overall distribution of fishing effort, remain important as the only source of information for establishing a fishery-wide impression of a fishery's impact on a marine mammal population. The advent of vessel monitoring systems (VMS) has improved the confidence of fishery managers in the reliability of that data (Gerritsen and Lordan 2011). Therefore, while observer programs remain the main tool for calculating by-catch rates across what is hoped to be a representative portion of the fishing effort expended, fishery logbooks remain important for establishing overall by-catch estimates, as was the case in chapters 4 to 6.

### *7.2.3 Marine mammal intelligence: learning and problem solving*

The main aim of chapters 3 and 4 was to develop and test the performance and efficacy of depredation and by-catch mitigation methods that were designed to either avoid or deter individual odontocetes that were engaged or had the potential to be engaged in operational interactions with fishing gear. Although avoidance strategies rely on the intelligence of humans to effectively ‘outsmart’ the marine mammals seeking to depredate from fishing gear, effective deterrence is dependent on having some understanding of the intelligence of the marine mammal involved.

One measure of intelligence in humans is the way, speed and extent to which individuals respond to novel information and new situations, which is exhibited through individual and group behaviours associated with perception, planning, problem solving and adaptation (e.g. Huphreys 1979; Sternberg and Salter 1982). In marine mammals, intelligence plays an important role in the development of complex social behaviours that underpin reproductive, foraging and predator avoidance strategies that have evolved to maximise individual and group success. Examples where intelligence plays an important role include male reproductive alliance affiliations in bottlenose dolphins *Tursiops truncatus* (Connor 2007), group strand-foraging in bottlenose dolphins (Duffy-Enchevarria et al. 2008, Fox and Young 2012) and diversion avoidance of predatory white sharks by Cape fur seals *Arctocephalus pusillus pusillus* (Martin and Hammerschlag 2012).

It is possible that marine mammals use their intelligence to circumvent depredation deterrence strategies used in fisheries, because they are able to solve problems and adapt to new situations. This has both positive and negative implications on the development, application and efficacy of deterrence strategies and technologies. In the context of this thesis, positive outcomes include common dolphins released from purse-seine nets in SA (see chapter 3) learning to avoid fishing activities due to the stress of being temporarily encircled and odontocetes unable to access an easy meal from a longline hook (see chapter 4) deciding to preferentially target wild fish. Similar situations have been reported in the Chilean Patagonian toothfish fishery (where physical deterrence has significantly mitigated depredation rates by sperm whales and killer whales; Moreno et al. 2008) and in drift gill-net fisheries (where captive individuals may have learned to avoid structures with which they have previously had negative interactions; Bowles and Anderson 2012). These outcomes are also likely to result in reduced by-catch of the marine mammals involved. In contrast, some marine mammals may decide to work harder to depredate fish from fishing gear, which places those individuals at increased risk of becoming by-caught. For example, Australian fur seals (*Arctocephalus pusillus doriferus*) and New Zealand fur seals (*Arctocephalus forsteri*) may learn to target holes in aquaculture cages or enter over above-water nets during periods of high swell (Robinson et al. 2008), while bottlenose dolphins may learn they can still effectively depredate fish from aquaculture cages despite the presence of acoustic harassment devices (AHDs; Lopez and Marino 2011). Therefore, each situation where marine mammal depredation and by-catch occurs should be viewed as unique and assessed on its individual characteristics.

The intelligence of marine mammals is the underlying basis for their behavioural adaptation from natural foraging behaviours to depredating from fishing gear. Unfortunately, this same ability allows marine mammals to adapt to the presence of deterrence structures on or near the fishing gear. As such, the effectiveness of many marine mammal deterrence strategies and technologies is likely to diminish over time, thus placing those individuals involved at increased risk of becoming by-caught. Therefore, managers and stakeholders should expect that, short of ceasing fishing activities altogether, effective mitigation of marine mammal depredation and by-catch will require sustained effort and commitment on their part, likely involving the simultaneous or consecutive implementation of a wide range of mitigation strategies and technologies in the long term, in order to combat the ability of intelligent marine mammals to rapidly adapt to changing circumstances and new situations (Campbell and Cornwall 2008; Gilman 2011).

### *7.2.4 Marine parks & spatial closures: limited protection*

Many marine mammal species and populations move over large distances in search of food. The most familiar examples are the baleen whales, travelling thousands of kilometres between winter coastal breeding grounds at lower latitudes and summer oceanic feeding grounds at higher latitudes (e.g. Mate et al. 2011). Many smaller marine mammal species also range over large distances, including the small odontocetes (e.g. spotted dolphins *Stenella attenuata* and spinner dolphins *Stenella longirostris*; Wade et al. 2006), and pinnipeds (e.g. New Zealand fur seals *Arctocephalus forsteri*; Baylis et al. 2008). Therefore, marine protected areas (MPAs) or analogous fishery closures may be too small to prevent the populations from being affected by threatening processes such as fishing, which may have direct impacts through by-catch, or indirect impacts through trophic competition or the destruction of prey habitat.

Despite earlier indications that Australian sea lions foraged over relatively short distances from their natal colony (Fowler et al. 2007), chapter 5 revealed that animals from some populations or breeding sites range over 100s of kilometres across the continental shelf adjacent to SA. Consequently, it was concluded that Great Australian Bight Marine Park (GABMP; the largest MPA along Australia's southern coastline) would not entirely prevent resident animals from being at risk of becoming by-caught in demersal gill-nets, because it extends only 21 kilometres south of the coastline at its widest point. Additionally, chapter 6 demonstrated that Australian sea lions continue to overlap extensively with demersal gill-net fishing in SA shelf waters, despite the 7.3 to 20.7 km exclusion zones around all 49 known breeding sites (AFMA 2012a). Similar MPA inadequacies have been highlighted elsewhere, with the Banks Peninsula Marine Mammal Sanctuary on the east coast of the South Island of New Zealand providing limited protection to Hectors dolphins from by-catch in gill-nets (Slooten et al. 2006). It was recently concluded that the area would need to be extended by 30 to 60 nautical miles to the north and to the south in order to provide adequate protection in order to reduce or prevent current levels of documented population decline (Slooten et al. 2006).

Despite the seemingly inadequate application of MPAs in some instances, there are some instances where they may be benefiting populations. For example, three much larger closures of areas approximating degrees of longitude and extending from the coastline out to the shelf edge have been implemented in SA, due to Australian sea lions by-catch limits being reached (AFMA 2011, 2012a, 2012b). These areas are likely to encompass most or all of the foraging ranges of several populations, some of which are very small, thus are likely to improve their conservation outlook. Similarly, predictive modelling indicates that the implementation of MPAs and fishery exclusion zones in the Mediterranean could have positive long term impacts on common dolphin populations even if they do not entirely prevent overlap (Piroddi et al. 2011). Therefore, adopting the precautionary approach by rapidly implementing MPAs to protect small or declining marine mammal populations from fishery impacts may be a wise first step, thus allowing additional time to characterise and quantify the interaction and to streamline protection measures to, where possible and appropriate, allow the two to coexist (e.g. Silva et al. 2012).

In recent times, it appears that some MPAs intended to protect marine mammal populations have been suitably planned and adequately implemented (e.g. Piroddi et al. 2011; Gormley et al.

2012). Nonetheless, one unintended consequence of such measures is the displacement of fishing effort to adjacent areas, outside the MPA, that may not have been fished previously, or as intensively. This may be of benefit for fisheries targeting less mobile, benthic species where a small MPA may assist in increasing birth rates inside, thus offsetting the increased harvesting pressure outside (e.g. Halpern et al. 2004). However, when the purpose of the MPA is to protect a population of highly mobile marine mammals that spend some portion of their time outside it, the displacement of fishing effort may cause impact on other elements of the life history or activities of the population. In essence, an MPA may result in a net increase in the risk of decline in the population, rather than a decrease as would be the aim. An example of this problem may be occurring in the Banks Peninsula Marine Mammal Sanctuary (BPMMS), which is designed to protect Hector's dolphins from becoming by-caught in demersal gill-nets (Rayment et al. 2010). The BPMMS extends 7.5 km offshore, although aerial surveys indicated that most of the population resided outside its offshore boundary during the winter months (up to 37 km offshore) where most of the fishing was concentrated (about 6 to 28 km offshore). This outcome suggests any gill-netting effort occurring in the area now occupied by the BPMMS prior to its proclamation may have been pushed into offshore areas, thus increasing the level of overlap there between the two. Therefore, resource and conservation managers need to carefully plan, implement and monitor the impact of MPAs to ensure their intended purpose does not change patterns of fishing in a way that has negative impacts on other species, or on other aspects of the species it is intended to protect.

## 7.3 Operational interactions: indicator of a wider problem?

The studies presented in this thesis, especially in chapters 3 to 6, suggest the full impact of operational interactions on marine mammals may be underestimated. This is because some drowned animals fall out of the fishing gear before being hauled to the surface while others escape with life threatening entanglements, without being detected and recorded by onboard observers. As such, actual levels of by-catch and entanglement are likely to be higher than conventional monitoring programs are able to reveal (e.g. Fowler et al. 1990; Warden and Murray 2011). There has been minimal effort to date to address this problem in any fishery. Observed operational interactions may also be the conspicuous element of a more cryptic ecological interaction, where individuals or populations of marine mammals are in trophic competition with commercial fisheries for access to a preferred fish species in limited supply.

Today's commercial fisheries have the ability to harvest and remove large quantities of fish from the marine environment and in doing so can decimate target fish species (Watson and Pauly 2001; Myres and Worm, 2003). As such, marine mammal populations targeting the same fish stocks must work harder to find fish because they will have become scarce, or must switch to other less nutritionally valuable fish species. Although the reasons remain unclear, it was speculated that overfishing for walleye pollock (*Theragra chalcogramma*) could have caused the significant decline of the southeast Alaska Steller sea lion (*Eumetopias jubatus*) population during the early 1990s (Calkins et al. 1998; Trites and Donnelly 2003; Hennen 2006; Trites et al. 2007). Similarly, overfishing of sardines may have caused the high rates of operational interaction reported in chapter 4 between common dolphins and the purse-seine fishery for sardines between 2004 and 2005, because competition to obtain the same resource was intensified. Although not quantitatively proven, the situation may have been occurred due to an erroneous tripling of the biomass estimate and thus the total allowable commercial catch (TACC) during the years 2001 to 2004 (Bernal 2006).

Continued trophic competition with fisheries may reduce the foraging success of marine mammals at an individual and population level, which over months may reduce survival rates of nutritionally dependent young and over years may reduce fecundity rates as adult females adapt their reproductive output to suit the less favourable foraging conditions (DeMaster et al. 2001; Lassalle et al. 2012). However, confirmed and quantitative examples of this effect on marine mammals are rare, principally due to two reasons. Firstly, it is difficult to establish the degree of competition for the same resource, because the diet of many species of marine mammal remains poorly or only partially understood and is likely to vary across time and space (Matthiopoulos et al. 2008). Secondly, obtaining reliable survival and fecundity data for predicting medium and long term population responses is expensive and logistically difficult, ranging to impossible for the more cryptic oceanic odontocetes. Despite these difficulties, the possibility of marine mammal population declines due to nutritional stress caused by trophic competition with fisheries should not be discounted as a major threat to the conservation of many marine mammal populations (Trites et al. 1997). This problem is likely to become more relevant as global fish stocks continue to decline at about 0.7 million tonnes each year (Watson and Pauly 2001), potentially resulting in intensified operational interaction and trophic competition between marine mammals and fisheries. Stakeholders and resource managers should thus consider the cumulative effects of all impacts on marine mammals, although fisheries having operational interactions with marine mammals may remain the focus of mitigation efforts, because they are conspicuous and the solutions, for the most part, are tractable. Therefore, all stakeholders should assume that even seemingly rare operational interactions between marine mammals and fisheries may indicate the occurrence of localised or widespread trophic competition, the impacts of which should be seriously considered when developing management responses aimed at mitigate impacts.

## 7.4 Marine mammal exploitation & conservation: can the future support both?

In a world where the human population continues to proliferate, increasing pressure is placed on our capacity to produce adequate amounts of food. While the terrestrial environment provides most of the food required by human populations, the marine environment has alleviated the pressure. Around 86 million tonnes of fish were estimated to have been harvested in 1997, although only 77 million tonnes were harvested in 2010, suggesting some level of overfishing and stock depletion (FAO 2012). Increased competition with other consumers such as marine mammals and fisheries is inevitable, with each being adversely impacted through increased levels of depredation and by-catch.

Stakeholders involved in addressing the problem of operational interactions can be categorised as either marine mammal conservationists or fishery proponents. Marine mammal conservationists have typically called for reductions in by-catch through tighter restrictions on fishing activities, arguing that efforts by fisheries to maximise profits often place marine mammal populations at risk of decline and increase the potential for the collapse of marine food webs (e.g. Myers and Worm 2003; Piroddi et al. 2011). Their view is that operational interactions between commercial fisheries and marine mammals can have only negative outcomes for the latter and that fisheries are doing little to stem the problem (e.g. Northridge and Hofman 1999; Read 2008). On the other hand, fishery proponents have traditionally called for reductions in depredation through culling of the offending marine mammals, arguing that the protection of marine mammals diminishes economic viability at a time when maintaining food supplies for human consumption is becoming particularly challenging (e.g. Blix et al. 1995; Jones 2008; Gerber et al. 2009). Achieving a balance between these opposing viewpoints has been difficult and largely unsuccessful, due mainly to concerns relating to food security in developed countries and to poverty in developing countries (e.g. Robards and Reeves 2011).

Unregulated hunting of marine mammals for products important during the industrial revolution of the 1700s and 1800s resulted in the collapse of many cetacean and pinniped populations (Hiller 1986; Ellis 1999). Although such practices now receive much more scrutiny, wilful naivety seems to have prevailed with regard to mitigating the negative impacts of operational interactions with commercial fisheries (Read 2008). Interestingly, the burden of proof still seems to be directed at those advocating precaution, rather than at those exploiting fish stocks (Agardy 2000). Amid this controversy, many marine mammal species and populations remain vulnerable to decline due to (i) life history characteristics that result in low fecundity rates (e.g. Australian sea lions: Hamer and Grayson 2012), (ii) small size due to lack of recovery after previous human induced declines (e.g. spotted dolphin and spinner dolphin by-catch: Wade et al. 2007) and (iii) the sheer intensity and scale of fishing activities today (e.g. in all oceans and most habitats; FAO 2012). As fishing effort continues to increase and targeted fish stocks continue to collapse (e.g. Myers and Worm 2003), these situations are likely to become more common.

Two human elements are likely to hinder progress towards marine mammal conservation. Firstly, varying cultural perception and values raise the question of whether whales should be hunted as a food source in a manner similar to many other marine species, or should be protected for biological and aesthetic reasons as they currently are under the International Convention on the Regulation of Whaling (ICRW) and the Convention on the International Trade of Endangered Species (CITES). Adopting the exploitative approach, Norway has interpreted the EBFM approach in a way that justifies sanctioned harvesting of marine mammals, as a valid means of enhancing production and sustainability of commercially targeted fish (NMFCA 2004). Japan also continues to harvest whales for 'scientific research' and also argues that 'whales eat fish' that are important for human consumption (MOFA 2012). Despite this claim, supporting evidence remains absent (Morissette et al. 2010) and there are alternative and non-lethal methods being developed to deter depredating whales (DSEWPaC 2010; Hamer and Childerhouse 2012). Secondly, fishery management in many developing countries are ineffective in mitigating the impacts of fishing on marine mammal populations and species, mainly due to a lack of political commitment and the necessary funding to build monitoring and compliance capacity. The artisanal and subsistence fishers of many developing countries opportunistically harvest marine mammals that become by-caught in their fishing gear, or are simply unable to avoid them with the comparatively primitive equipment in use (Alder et al. 2010; Robards and Reeves 2011). Additionally, many fishers are likely to be unaware of the impact of their activities on marine mammal populations, due to limited flow of information and education. In contrast, developed countries such as Australia have implemented processes to formally scrutinise and manage the impact of fishing activities on marine mammals through the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act), using the *Guidelines for the Ecologically Sustainable Management of Fisheries* (DEWR 2007). The presence of fishery compliance officers at sea and in ports along with electronic vessel monitoring systems (VMS) ensure that the regulations implemented pursuant to the EPBC Act are effectively implemented and understood. However, despite a 2005 ministerial direction pursuant to the *Fisheries Administration Act 1991* that overfishing in Australian waters should be immediately halted, assessments of 101 commercial fisheries in 2009 revealed that overfishing continued in 15 of them and that insufficient data to reliably determine the situation was the case for another 30 (Wilson et al. 2009). Chapter 4 and literature subsequent to the results presented in chapters 5 and 6 (e.g. AFMA 2010, 2012b) indicate that the impact of fisheries on marine mammal populations may continue, despite the legislative frameworks in place.

To some extent, the development of marine protected area networks and of shore-based and coastal aquaculture activities are designed to improve the sustainability of fishing, by providing havens for juveniles and egg producers, and by deflecting demand and thus pressure from wild fisheries, respectively (FAO 2012). However, some argue that these efforts are still unlikely to prevent the depletion of many more commercially targeted wild fish populations (e.g. Kurlansky 1998; Clover 2008). Nonetheless, the United Nations (UN) recommended that fishing nations should strive to achieve ecosystem based fishery management (EBFM) by 2010, in an attempt to address this problem (Ward et al. 2002; FAO 2012). Many of the Regional Fishery Management Organisations (RFMOs) targeting tunas claim to have addressed non-target by-catch, although implementation seems to have been difficult because the definitions and principles remain unclear, the influence of a dynamic ecosystem on comparatively rigid fishery management frameworks are complex, and the stakeholders almost always have different agendas and goals (Stump 2009; Gilman 2011; Hamer et al. 2012). Therefore, those more able members of RFMOs have an opportunity and a responsibility to assist in capacity building with less able members, in a bid to protect fish stocks, marine mammals and the ecosystems generally, focusing on biological rather than jurisdictional boundaries.

This thesis presents an extensive body of work on the characterisation of operational interactions between marine mammals and commercial fisheries and on attempts to mitigate the adverse impacts on one or both parties. Each study provides evidence that all stakeholders can, when there is sufficient will, cooperate to solve these complex problems. Recent additions to the published literature, aside from those associated with this thesis, indicate that similar efforts are also being made in other regions (e.g. Zollett and Read 2006; Moreno et al. 2008; Rabearisoa et al. 2012). Although these efforts provide promise in developed countries, the future for marine mammal populations in waters around many developing countries remains bleak, because the need to mitigate the adverse impact of fishing activities is yet to receive serious attention (e.g. Read 2008; Robards and Reeves 2011). Additionally, while there is increasing pressure to exploit the remaining commercially attractive fish stocks in the world's oceans for human consumption, it is likely that the adverse impact of fishing activities on marine mammal populations will also continue to increase. Therefore, if marine mammals and commercial fisheries are to coexist indefinitely, there is a need for developed countries to prioritise efforts toward mitigating operational interactions in their own jurisdiction and to facilitate capacity building in developing countries.

## 7.5 Synthesis & future directions

In summary, this thesis has addressed the three original aims. Firstly, chapter 2 reviewed a major fishing method in the South Pacific region that has operational interactions with odontocetes, and which is likely to become an increasing problem on a global scale. Secondly, chapters 3 to 6 characterised the nature and extent of operational interactions of small and large odontocetes and pinnipeds with three major fisheries in the Oceania region. Thirdly, chapter 3 and 4 explored the efficacy of different approaches to mitigating operational interactions by using physical and psychological deterrence and changes to fisher behaviour and to fishing gear.

The findings presented here demonstrate that, where there is sufficient will, cooperation and capacity, it is possible to make considerable inroads into characterising and mitigating operational interactions between a marine mammal population and a commercial fishery. In chapters 4 to 6 where by-catch was the primary concern, the commercial fisheries involved were exposed to considerable commercial risk by allowing observers to accompany their vessels to undertake the research. It should be pointed out that although each fishery was faced with the prospect of future mandated observer coverage, particular licence holders carried observers voluntarily to assist with the research. The fishery-wide commercial risk was realised to some extent in all three fisheries; in chapter 4 with a mandated increase in observer coverage and in changes to fishing practices during the latter half of the study period (Hamer et al. 2008), subsequent to chapter 5 with proposed extensions to the GABMP (DSEWPaC 2012), and subsequent to chapter 6 with increased observer coverage, by-catch limits and associated area closures (AFMA 2010, 2012b). Given that depredation was also a concern in chapter 3, the aim was to achieve positive outcomes for both the marine mammals and the fishery involved. In the longer term, it is hoped this will be realised through an ongoing project or projects that carry on the development of the two physical and psychological deterrence devices and their derivatives. However, there may be a need to identify regions where the level of operational interactions between odontocetes and pelagic longline fisheries are known to be high, so that a situation similar to that reported in chapter 4 (where depredation and by-catch events were sufficiently rare to prevent meaningful statistical analyses from being carried out) can be avoided. A similar outcome was recently reported in a demersal longline fishery (see Moreno et al. 2008), which provides hope, although it is likely that the odontocetes involved may circumvent the deterrence devices because they are intelligent, necessitating further development and refinement. Nonetheless, these studies aim for the best possible outcome; a 'win-win' situation for both the marine mammals and the fishery involved, where less caught fish are damaged resulting in increased profits for the fishery and where less marine mammals are by-caught resulting in decreased risk to their conservation and welfare.

In extreme cases, the restrictions imposed on a fishery to address and mitigate operational interactions with marine mammals can be severe. Unfortunately, gear modifications (such as in chapters 3 and 4) and changes to fisher behaviour (such as in chapter 4) cannot be applied to all fishing methods, thus leaving changes in gear type, reductions in by-catch limits and reductions in overlap through spatial closures as the only feasible options. The management response to the findings presented in chapters 5 and 6 demonstrate how extensive spatial restrictions can be, where unsustainable levels of Australian sea lion by-catch in demersal gill-nets have been mitigated by closing about 70% of the area previously available to the fishery. Several licence holders in the fishery have since voiced their concern about the future viability of the demersal gill-net fishery under the new management arrangements; a transition to longlines may alleviate their concerns. Despite the legitimate reasons for attempting to mitigate operational interactions between marine mammals and fisheries, increasing concerns over food security and production to feed the burgeoning human population are likely to cause increased tension between exploiters of marine resources (along with their managers) and marine mammal conservationists.

This thesis has laid a sound case for continued development and use of physical and psychological deterrence technologies, based on the series of promising results presented. This approach is relatively new, with efforts to mitigate odontocete depredation and by-catch over the last decade focusing on acoustic deterrence technologies. However, they have been limited in their application due to the large size of their transponders and batteries, plus the limited understanding of the frequency and magnitude of the sound they need to produce (e.g. Dawson et al. 2012). Should this imitation be resolved through increased efficiencies and other avenues, it may be possible to integrate acoustics with some of elements developed in chapter 3. For example, the branchline attachment and tension triggering mechanisms may be used to switch on an acoustic deterrence device and then position it next to the caught fish. The advantage of this approach is that less energy and sound would be required to deter a depredating whale because the device would be in close proximity to the caught fish; a situation yet to be achieved in any other acoustic deterrence device. Similar innovations, where acoustic deterrence is integrated with trigger mechanisms, could be applied to other fishing gear types. In demersal gill-net fisheries, for example, small acoustic deterrence devices with buoyant outer cases could replace the small floats attached at short distances along the floatline. A tension activated switch, possibly tethered down the side of the net, would switch on the device after a fish is caught in the nearby meshes, thus deterring depredating pinnipeds and mitigating catch damage and pinniped by-catch.

Some cases of operational interaction between marine mammals and fisheries will continue to require strategies that are detrimental to the fishery involved. This is becoming increasingly true as communities, especially in developed countries, demand fish products to be obtained using techniques that have minimal impact on the broader marine environment. However, in a world where it is becoming more and more difficult to obtain food sustainably while still meeting food demand, there is likely to be more resistance to such practices in the future. Nonetheless, the importance of continued commitment by all stakeholders towards improvement and development in this area of research cannot be overstated. Therefore, the challenge into the future is to find ways to effectively mitigate operational interactions between marine mammals and fisheries that minimise the conservation and welfare impact on the marine mammal populations involved, while also minimising the economic impact on the fisheries involved.

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